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REVIEW OF EFFECT OF SOIL ON RADIONUCLIDE UPTAKE BY PLANTS

by

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ABSTRACT

This review was undertaken to improve the understanding of, and to compile the available data concerning, the transfer of uranium (U), thorium (Th) and lead (Pb) from soils to plants. The emphasis of the review was on the absorption of these elements from the soil by plant roots, and the mechanisms underlying this process were outlined. The behaviour of U, Th and Pb in soils and plants was discussed with illustration by data from the literature. An extensive compilation of plant/soil concentration ratios (CR) was completed and the most relevant data for Canadian nuclear facilities were selected. Very few data were found for edible plants and these did not represent the range of soil types found near Canadian nuclear facilities. Recommendations of the most fruitful research directions were made.

RÉSUMÉ

On a entrepris cet examen pour mieux comprendre le transfert de l'uranium (U), du thorium (Th) et du plomb (Pb) des sols aux plantes et rassembler les renseignements qui existent sur ce sujet. Le centre de l'examen a été l'absorption des éléments par les racines des plantes à partir du sol et les mécanismes à la base de ce processus ont été décrits sommairement. On a examiné le comportement de l'U, du Th et du Pb dans les sols et plantes par illustration avec des renseignements provenant de la bibliographie. On a rassemblé en profondeur les rapports de concentration (CR) plantes/sols et choisi les renseignements se rapportant le plus aux installations nucléaires canadiennes. On fait des recommandations quant aux voies de recherche les plus fructueuses.

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1. INTRODUCTION

1.1 The Role of the Soil-to-Plant Pathway

Man, like all other forms of life, is simply an organized set of atoms, molecules and compounds. Because these materials originally all come from the environment, life forms are predisposed to absorb materials from the environment. The unfortunate aspect of this is that potentially harmful materials, such as radionuclides, cannot be excluded from the absorption pathways. Man is susceptible to the absorption of radionuclides through many pathways, including food and water ingestion, air intake and direct contact with the skin. In the case of radionuclides, the potency of each pathway varies with the chemical forms and the sources of the radionuclides. However, the largest intake of mass by people is undoubtedly by way of food ingestion. Thus, for many radionuclides, especially those that are not volatile and are only sparingly soluble, the most potent pathway is through food.

Food, derived from either animal or plant products, can also receive radionuclides from many pathways. For animals, the largest intake of mass is again through food intake, specifically the consumption of plants. Plants acquire the greatest amount of their mass through the fixation of carbon dioxide (CO_2) from the atmosphere, with the soil providing most of the remainder. Thus, the pathway from soils to plants becomes a fundamental step in many of the pathways to man, and often is critical in the more potent pathways.

The importance of the soil-to-plant pathway is implicitly acknovledged by the fact that it is almost always included in nuclear environmental safety assessment models. The traditional food chain models vere essentially linear, multiplicative models (Baker et al., 1976). A two-fold increase in the simulated transfer of radionuclides from soils to plants lead directly to a two-fold increase in the estimated dose to man. Even on a larger assessment scale, the soil-to-plant transfer emerges as a significant pathway. In the second interim assessment of the Canadian concept for nuclear fuel vaste disposal

(Wuschke et al., 1985), sensitivity analysis of the entire system, from a source in a deep underground vault to man living on the surface, found the soil-to-plant transfer to be a critical pathway. Specifically, it was found that variation in the estimated dose to man was more closely related to variation in the soil-to-plant transfer than to variation in most other pathways (Walker, 1985).

The soil-to-plant transfer is a major and fundamental step in the passage of radionuclides from abiotic, physically driven media into living organisms. Although not usually thought of as a bioconcentrating process, plants do accumulate higher concentrations of some radionuclides than are present in the corresponding soil. Baes et al. (1984) revieved the data on the transfer of 62 elements from soils to plants and, of these, nine were accumulated to concentrations (on a dry weight basis) above those of the corresponding soil.

1.2 Outline of Basic Processes of Soil-to-Plant Transfer

The study of the relationship between soils and plants has evolved to a discipline in itself, encompassing biological, chemical, mathematical and physical sciences. As such, several excellent reviews have been published (Clarkson and Hanson, 1980; Barber et al., 1984; Carson, 1974). The intent in this chapter is only to provide a brief overview of the subject, leading to a connection between this state-of-the-art understanding of the processes and the generalization of these processes into one parameter, the concentration ratio.

Soil-plant interactions can best be vieved on three scales. The macro-scale view is the plant as a hydraulic conduit or pipeline for water stored in the soil to travel upward and join water dispersed in the atmosphere. The meso-scale view is the plant as an organism that attempts to regulate the uptake and the loss of materials from and to the environment. The micro-scale view is of the plant root surface as it actively and passively modifies the soil constituents.

On the macro-scale, plants must conduct water from the soil to the atmosphere in order: to draw mineral nutrients to the roots; to replace water that inevitably diffuses out of plants by the same pathway by which CO_2 diffuses in; and to cool the leaf surfaces. All of the water that passes through the plant must also pass through the soil around the roots. Hence, all of the materials dissolved in this stream of water have been brought to the root surface. A small flow, for example by a plant that conserves moisture, will mean few materials are brought to the root surface. Thus, the water flow through the plant, or the transpiration stream, is a key driving force in the absorption of materials from the soil.

The transpiration flow and growth rate of the plant are mechanistically linked. Plants grow faster if they can absorb and fix CO_2 from the atmosphere more quickly. The CO_2 is absorbed through the stomates or pores in the leaves. If the stomates remain open longer to allow absorption of more CO_2, an inevitable consequence will be greater loss of water vapor. Thus, a reasonable generalization is that plants that grow faster, due to either genetic or environmental factors, will draw more water from the soil. This will tend to increase the total amount of materials absorbed from the soil. In terms of concentration in the plant, the effect is not as clear because plants that grow faster may also grow bigger, causing "biological" dilution of the absorbed materials to lower amounts-per-unit-of-plant-weight.

On the meso-scale, plants have developed to discriminate at least some of the materials drawn to the root surface by the transpiration stream. The plant accomplishes this by existing as a two-tiered organism. The outer tier, or apoplast, is made up of the cell walls and extends from the root tips to the shoot tips. The cell walls are porous and inter-connected, much like the concrete of a concrete-block building. The plant has little control over materials that diffuse or flow into the apoplast. Within the apoplast is the inner tier, or symplast. The symplast includes all of the biologically active parts of the plant and is bounded by membranes. The symplast is also interconnected and continuous from cell to cell throughout the plant.

The membranes surrounding the symplast are actively regulated by the plant. Metabolic energy is used to discriminate among materials and either accelerate or restrict their uptake. The plant has evolved to be able to accelerate the uptake of growth-essential elements and to reduce the uptake of toxic materials. Plants have not evolved mechanisms specifically to deal with most radionuclides. Some radionuclides may be mistakenly identified by the membrane to be essential, or to be toxic, whereas others may be free to passively diffuse through the membrane.

The same types of regulatory mechanisms occur throughout the symplast, controlling passage of materials in and out of the plant's fluid conduits, the phloem and the zylem. Some materials may be irreversibly deposited in certain tissues, whereas others may remain relatively mobile. For growth-essential elements, plants can actively conserve them by mobilizing them from less-active or dying tissues and routing them to active tissues. Some radionuclides may also be handled in this manner. Conversely, materials that are not closely regulated in the plant may deposit anywhere in the plant, depending upon their solubility. Relatively soluble materials are often drawn by the evapotranspiration stream to leaves and leaf margins.

On the micro-scale, plant roots have evolved mechanisms to modify the soil and create a more favorable nutrient environment. There are three major categories of these mechanisms; stimulation of soil microbial growth, side effects or byproducts of the activities of the root membranes, and excretion of highly selective enzymes into the soil. Microbial activity is stimulated passively by root exudates and sloughed-off cell materials. Active or synergistic stimulation of microbial activity also occurs. The consequences of this microbial activity include changes in acidity (pH), changes in oxygen (O_2) and CO_2 concentrations, and fairly specific dissolution activity on soil minerals. Furthermore, the combination of microbial biomass and root exudates can create a "mucigel" which effectively increases the surface contact between the roots and the soil.

At least two effects resulting from the activities of the root membranes can occur. One is that the root will exude hydrogen cations or bicarbonate anions to maintain charge balance. For example, if the plant is absorbing more cations than anions in an effort to satisfy nutritional requirements, it will then exude hydrogen ions to balance the charge. This will cause the pH of the soil adjacent to the root, the rhizosphere, to decrease. The other side effect is that certain materials will be depleted or accumulated in the rhizosphere if the plant uptake rate differs markedly from the rate of flow to the root of that material in the evapotranspiration stream. In extreme cases, solubility limits can be exceeded and the materials may precipitate in a cylinder around the root.

It has been recently learned that plants can exude chemicals with fairly specific actions on the soil. An example would be chelate-like substances that form complexes with elements in the soil, resulting in increased solubility, and hence increased likelihood of absorption by the plant.

All of these micro-scale processes can impact upon the uptake of radionuclides. For example, uranium (U) often accumulates in or on roots and a common hypothesis is that it precipitates on the root surface or in the apoplast.

This entire discussion serves to demonstrate the complexity of soil-plant relations and processes. Other variables such as plant-species, climatic and soil effects will express themselves through these processes in various ways. Attempts to model the processes in a mechanistic manner (Cushman, 1979; Baldwin et al., 1973; Smith, 1976; Itoh and Barber, 1983) have not evolved to a broadly-applicable level. Because of this, the subject is often treated at an empirical level. The next section discusses one of these empirical approaches. Without doubt, the usefulness of the empirical approaches will be improved if they bear some relationship to the basic processes in soil-plant relations, specifically:

1) the solubility of materials in the soil,
2) the discriminatory activities by the plant, and
3) the flow rate of the evapotranspiration stream.

1.3 The Concentration Ratio (CR) Model

The most common model used to describe the transfer of radionuclides from soils to plants is the CR model. It is an entirely empirical approach, essentially saying that the concentration in a plant is a simple, linear function of the concentration in the soil.

$$CR = \frac{\text{concentration in the plant}}{\text{concentration in the soil}}$$

It should be clear from the previous section that CR values are dependent upon a great many factors. Conceivably, every possible combination of plant, soil and environmental condition will have a unique CR for each radionuclide element. However, as with any empirical quantity, the average over several combinations of conditions is usually sought, and is sufficiently valid.

There is an inconsistency in the literature in the way CR values are expressed. Various alternative terms including "transfer factor (TF)", "concentration factor (CF)", and "coefficient of biological absorption (CBA)" have been coined to deal with this. One problem is that fresh, dry or ash weights are used to express plant concentrations. Soil concentrations are usually expressed on a dry-weight basis, although some authors may use an ash-weight basis. Conversion between the modes of expression is possible. Generalized dry matter contents of 25% (dry/fresh weight) and ash contents of 1-5% (ash/dry weight) for plants can be used.

CR values are inherently variable, typically spanning three to five orders of magnitude. They are lognormally distributed (Sheppard, 1986) and thus, all averages should be computed as geometric means. We define geometric mean (GM) and geometric standard deviation (GSD) as the arithmetic mean and standard deviation of data transformed to the natural logarithm scale. We present e^{GM} as the mean values.

The variability in CR values reflects the range in measurement techniques, plant types, soil types and climates used in the experiments. Measurement techniques are, unfortunately, quite variable. One of the first sets of CR values was computed as ratios of "typical concentrations" in soils and plants as compiled by Ng et al. (1968). These concentrations were unpaired and came from measurements made throughout the world. These CR values were broadly used while researchers conducted experiments to obtain paired plant and soil data. Various techniques were used in this effort to generate better CR values. Some CR experiments used solution culture where the plants were grown in a nutrient solution contaminated with the radionuclide, but not including any soil. These data are of little use and have been removed from most compendiums of CR values. Experiments are also commonly conducted with plants grown in pots, often in controlled environment facilities. CR values from these experiments are usually higher than those measured in more realistic field settings. The most reliable data come from long-term experiments in realistic outdoor settings in which soils are contaminated either artificially, or by some industrial process or accident. The International Union of Radioecologists is the only group to have established a comprehensive experimental program to measure CR values under a broad range of real-world settings (IUR, 1984).

Variations in methodology exist among even the best experimental approaches. For example, experiments designed to emulate atmospheric deposition following a reactor accident apply the radionuclide to the soil surface and measure plant uptake immediately. In contrast, experiments designed to emulate waste repositories may mix the radionuclide uniformly in the soil and allow it to react with the soil for some time before plants are grown. These experimental differences can lead to marked differences in the estimated CR values. A useful example involves technetium (Tc). This element reacts slowly with soil resulting in less-soluble forms with time. This process continues for several months before a quasi-equilibrium or steady state is reached. Plants grown immediately after contamination with Tc can remove essentially all of the Tc from the soil (Sheppard et

al., 1983), whereas plants grown on soils contaminated for several years by Tc absorb a relatively small proportion of the Tc in the soil (Garten et al., 1986).

The CR model of plant accumulation of radionuclides implies the assumption that concentrations in plants are linearly correlated to those in the soil. Within a narrow set of circumstances, for example within one experiment, this assumption may by valid. Sheppard and Evenden (198X) examined this linearity for five elements applied to soils in concentrations that spanned up to eight orders of magnitude. In their experiment, the concentrations in the plants were approximately, linearly related to those in the soil, in some cases over the full range in soil concentrations. Errors associated with using CR values to represent the data were less than two-fold for the elements iodine, selenium, cesium and lead (Pb). Errors of over ten-fold would have resulted from using CR values to describe plant accumulation of U. In the case of U, the plant concentrations were linearly related to the concentration of U in the soil pore water. This illustrates the importance of solubility in the soil on uptake of plants, as shown by Sheppard (1985).

The correlation of concentrations in plants to those in soils is intuitively expected. However, when data from many environmental settings are combined, it is common to find no statistically significant correlation (Amiro and Sheppard, 1987). Sheppard and Evenden (unpublished data) surveyed concentrations in plants and soils of 29 elements for one native plant species throughout the range of that species in Canada. Concentrations in the plants were correlated to those in the soil only for about one third of the elements surveyed. This probably reflects the effects of all the other environmental influences on plant growth and performance. These effects will tend to lessen the importance of concentration in the soil on the accumulation of elements by the plants. The range of soil concentrations found by Sheppard and Evenden usually spanned one to three orders of magnitude. It may be that significant correlations would have been found if a broader range of soil concentrations had been sampled. The consequences of the lack of correlation does not

negate the value of the CR model. Plants accumulate different elements to markedly different extents, and the CR model is useful to model these differences. The lack of correlation must be considered as another source of variability in CR values. It demonstrates the complexity of soil-plant systems, especially when compared across the range of natural environmental conditions, and it illustrates the futility of applying more complex, mechanistic models to describe plant uptake in a generic sense. Another implication of the lack of correlation is that CR values computed as done originally from the data of Ng et al. (1968) from unpaired observations of plant and soil concentrations may well be valid and useful data. If plant and soil concentrations are not correlated, then the ratio of the means will be equivalent to the mean of ratios.

Several major compendiums of CR values have been published (IUR 1984; Baes et al., 1984; Coughtrey et al., 1985 and Ng et al., 1982). That prepared by Baes et al. includes the largest number of elements and is the only one to interrelate CR values between elements and between plant types in a systematic manner. They made use of correlations to other parameters to compensate for paucity of data for some elements. They also related the estimated CR for the elements to their position in the periodic table of the elements. Given that the mean CR values for each element are intended to be representative of all possible situations, it is reasonable to expect them to reflect the basic chemical characteristics of the elements. For example, Baes et al. found a consistent pattern among the CR values of the transition elements when they were plotted by periodic-table Group. The elements of Group 7b, namely manganese, technetium and rhenium, had the highest CR values for their respective Period. Elements on either side in the same Period had progressively lower CR values. It is our view that such systematic variation in CR values should be recognized and used to confirm values of CR for elements where few data have been reported.

1.4 The Objectives of This Review

The objectives of this review are:

1) To review literature on the specific interactions of U, Th and Pb with soils and plants in order to identify factors that can be expected to have marked effects on the corresponding CR values.

2) To compile CR values for U, thorium (Th) and Pb with particular emphasis on those values most representative of conditions around Canadian nuclear facilities.

3) To identify weaknesses and strengths in the compiled data set, and to identify the most fruitful research direction.

2. ENVIRONMENTAL BEHAVIOUR OF URANIUM, THORIUM AND LEAD

2.1 Reaction with Soil

The elements U, Th and Pb each react with soil solids in some way that retards their migration through the soil. The degree of retardation differs among the elements and from soil to soil. Thus, an understanding of the types of reactions and the factors that control them is important.

One of the first points to establish is whether the various isotopes of each element behave similarly in terms of chemical reactions. Fractionation of isotopes by environmental processes can be significant for low-atomic-weight elements (Mariotti et al., 1980). Reports of differential behaviour of higher-atomic-weight elements can be found, but as in the case of Paul and Pillai (1986), this can usually be traced to chemically and/or physically different sources for each of the isotopes. The similarity in behaviour of isotopes in natural settings has been demonstrated for $^{234}U/^{238}U$ and $^{230}Th/^{232}Th$ by Evans and Eriksson (1983) and for ^{210}Pb/stable Pb by Persson (1974) and Sharma and Kalin (1982). Thus, information on the behaviour of one isotope of U, Th or Pb in the environment is applicable to all isotopes of these elements.

2.1.1 An Overview of the Properties and Classification of Soils

The soil may be the most complex environment on Earth. Its solid mineral structure resembles geological materials. Its organic residue shares characteristics with all other natural organic residues. Its water chemistry resembles that of the oceans. Its gaseous component or atmosphere ranges from fresh to as foul as is environmentally possible. Its living component, including microorganisms and invertebrates, supplies the soil with enzymatic activities and energy budgets almost comparable to an organism. Soil is not static but develops with time through the dynamic interaction of these components. The original mineral structures slowly dissolve; but new mineral structures are formed either by transformation (discongruent dissolution) of the original structures, or by crystallization and accretion of new materials. The organic residues are attacked by the soil organisms and slowly decay; but then new organic residues are added by continuous death of plant and soil organisms. The soil water is diluted by rainfall; enriched by soluble fractions of the mineral and organic soil solids; and further enriched when evaporation concentrates the water. The soil atmosphere is pumped in and out of the soil by thermally- and hydraulically-induced pressures. It is depleted of O_2 as it passes downward, and it becomes substantially enriched in CO_2. When the O_2 supply becomes especially limiting, then gases typical of anaerobiosis, such as methane and hydrogen sulphide, become prevalent in the soil atmosphere. The living component of the soil interacts with all parts of the soil, but undergoes distinct cycles with time, related to the seasonality of temperature and rainfall.

The net effect of these soil processes is the development of a solum or weathered blanket of soil. The characteristics of the solum change with depth, and this "profile" with depth is used as a means to classify soils. Soils in Canada are all relatively young and their development is sloved by low annual temperatures. All of Canada's nuclear installations are in regions where there is a surplus of rainfall relative to evapotranspiration, with the result that a net annual downvard movement of water occurs. The low temperature and net

annual downward movement of water effectively limit the number of soil classes we need to consider in an assessment of soils relevant to nuclear installations. The relevant soil classes are described in Table 2.1, and the features of special importance with respect to the mobility of U, Th and Pb are identified.

There are a few chemical characteristics of soils that are especially important. One is the ability of soil solids to adsorb materials from soil solution. Soil solids have a net negative charge and thus must have a sufficient quantity of positively-charged ions, or cations, adsorbed to their surfaces to balance the charge. These cations can exchange with others in solution. The amount of cations held in this way is called the cation exchange capacity. Obviously, the cation exchange capacity will vary with the mineralogy and particle size of the soil solids. These soil solids also have a limited number of positively-charged sites which attract negatively-charged ions, or anions.

Few, if any, solid surfaces in soil represent regular, crystalline structures. The surfaces are usually altered in some way, often because of coatings of organic materials or iron oxides and hydroxides. These surfaces lend a uniqueness to each soil in terms of chemical reactivity.

Another component of many Canadian soils worthy of mention is the carbonate content. Carbonate minerals are eventually weathered out of soils, but while they are present, they can have a profound influence. Their major effect is on the pH of the soil solution. While carbonate minerals are present, the pH will be no less than 6.5, and will usually be 8.0 to 8.5.

Organic soils are common in Canada and differ markedly from mineral soils. Like mineral soils, organic soils also tend to have a net negative charge. However, this charge is dependent upon the pH of the soil, the botanical source of the organic soil and the degree of decomposition or "humification". Many organic soils are acidic, but this is not a universal characteristic, because organic soils formed

Table 2.1 Description of soil types, based on the Canadian system of soil classification, relevant to this review[1]

Soil order	Description[2]	Characteristics relevant to this review
Brunisolic	Brunisolic soils occur in all soil temperature regimes, and under a broad spectrum of vegetation types although forest cover predominates. They have brownish B horizons (Bm). The four great groups - Melanic, Eutric, Sombric and Dystric Brunisols - are separated on the basis of thickness of Ah horizons and soil reaction.	Often considered as intermediate in terms of development, these soils are progressing toward luvisol or podzol characteristics. Thus, this order includes a broad range of chemical characteristics. Those soils developed on carbonate parent materials will contain carbonate minerals in the surface horizons.
Gleysolic	Gleysolic soils develop under conditions of periodic or prolonged saturation with water. They occur under grass, forest or heath vegetation in boreal to subarctic, subaquic to peraquic soil climates. In all soil climatic regions, they occur in poorly drained depressions, but in humid regions they may occur on slopes and uplands. Gleysolic soils are recognized by mottles and dull colors within 50 cm of the surface that are caused by chemically-reducing conditions. Three great groups - Humic Gleysol, Gleysol, and Luvic Gleysol - are separated on the basis of the thickness of Ah horizons and the presence of Bt horizons.	The saturation with water limits oxygen supply to these soils, resulting in a chemically-reduced environment. Leaching is often limited since water flow is restricted. Organic residues can accumulate on the surface from the growth of sedges, etc.
Luvisolic	Luvisolic soils occur under deciduous, coniferous and mixed forest vegetation, and soil climates ranging from mesic to subarctic and humid to sub-humid. They have grayish strongly eluviated A horizons (Ae) and brownish illuvial B horizons (Bt) enriched with translocated silicate clay. Typical developmentment is on	The parent materials usually contain carbonate. The clay particles move from the surface layers only after the carbonates have been dissolved out of the surface layers. Soil pH is 6-8.

Soil order	Description[2]	Characteristics relevant to this review
	well-to imperfectly-drained sites in medium- to fine-textured, calcareous, parent material. The two great groups - Gray Brown Luvisol and Gray Luvisol - are separated on the basis of the nature of the mineral-organic surface horizon and the associated soil climate.	
Organic	Organic soils are composed dominantly of organic materials. They occur widely in perhumid to peraquic soil moisture regimes and soil temperatures warmer than subarctic. Those developed from hydrophytic plant materials are differentiated by the degree of decomposition of their organic materials. These are the Fibrisol, Mesisol and Humisol great groups. Others developed from forest litter, overlying rock or fragmental material, are Folisols and are seldom saturated with water.	Many of these soils are periodically saturated with water resulting in a chemically-reduced environment. The pore waters contain relatively high concentrations of soluble organic materials. The solid materials can effectively sorb dissolved minerals.
Podzolic	Podzolic soils mostly occur under forest or heath vegtation in boreal of cryoboreal, humid soil climates. They usually have organic surface horizons composed of forest litter, that are underlain by light-colored eluvial horizons. They have brownish B horizons enriched with amorphous materials composed mainly of organic matter combined with aluminum and iron. The parent materials mostly are coarse textured and acidic. Three great groups - Humo-Ferric Podzol, Ferro-Humic Podzol and Humic Podzol - recognize increasing degrees	These soils are acidic and sandy with low sorbing capacities.

Soil order	Description[2]	Characteristics relevant to this review
	of accumulation of carbon, aluminum and iron and broadly reflect increased humidity of the soil environment.	
Regosolic	Regosolic soils are weakly developed; some may have a dark colored surface horizon (Ah) or buried horizon (Ahb) but all lack development of other genetic horizons. Their weak development may be due to youthfulness or instability of the deposit, to the resistant nature of the material or to dry or cold climatic conditions. They occur under all soil temperature regimes and a broad spectrum of vegetation. The two great groups - Regosol and Humic Regosol - are differentiated on the basis of A horizon thickness.	These soils are essentially the unconsolidated mineral materials left by glaciation or other geomorphic processes. They, like the Brunisols, may encompass a broad range of characteristics.

[1] The soil orders Chernozemic (prairie soils), Cryosolic (permafrost soils) and Solonetzic (arid, saline soils) were omitted because they have little relevance to nuclear installations in Canada.

[2] Taken from Bentley (1979), the key soil layers or horizons are termed A, B and C with subscripts (eg. Ae, Ah, Ahb, Bm, Bt) to define specific characteristics.

in contact with calcareous mineral soils will have pH values as high as 7.0 or 8.0 . The chemistry of organic soils is very complex and seems to be unique for each soil.

2.1.2 The Behaviour of U in Soil

The behaviour of U in soil and in the geological environment has been reviewed a number of times (Sheppard, 1980; Harmsen and de Haan, 1980; Paquette and Lemire, 1981; Ollila, 1985; Adams et al., 1959). We will list the soil properties these authors found to be important and will illustrate several of the specific points with the findings from original papers. The valence state of U in soils is either U^{4+} or U^{6+}. The U^{4+} state would be expected in strongly-reducing environments, and can be formed during the oxidation of organic matter or iron (Fe) in soils. The U^{4+} state would occur as solid UO_2 (very low solubility) or UOH^{3+}, which would be strongly adsorbed. Thus, the U^{4+} state is very immobile and not common in aerated soils. The U^{6+} state typical of oxidizing environments will invariably be found associated with oxygen as the uranyl ion UO_2^{2+}. This ion is strongly adsorbed by soils, but can form stable complexes with many ligands, notably CO_3^{2-} and organic complexants. At higher concentrations of these ligands, the complexes tend to have lower positive or even negative charges. Thus, these complexes increase the mobility of U. The mobility of U can also be influenced by the presence of colloids, especially when the colloids include insoluble U oxides that would otherwise be expected to be immobile (Giblin et al., 1981). Microorganisms can remove U from solution, largely by adsorption onto their tissue surfaces (Horikoshi et al., 1981); but it is not clear what net impact this may have on mobility.

The characteristics of U we have outlined indicate that soil properties associated with reduced aeration (i.e. water saturation, high biological or chemical O_2 demand), organic matter content (i.e. soil genesis), carbonate content (i.e. pH, parent material, degree of weathering) and cation exchange capacity (i.e. texture, clay content, organic matter content, pH) are likely to be important.

The effect of decreased aeration on the mobility and plant-availability of U was illustrated by Sheppard and Evenden (1985). They placed U in 2-cm-deep bands either above, at, or below the water table in separate lysimeters. After one summer, the soils were analysed with depth and in each case the U tended to move upward in response to the evaporative flux of water (Figure 1). However, twice as much U was able to migrate when it was originally placed above the water table than when placed at or below the water table. Since the water flux was essentially the same in each case, this effect can be clearly linked to the decreased mobility of U in reducing environments.

The interaction of U with organic materials, including organic soils, is of special importance. Peat soils have a substantial capacity to immobilize U. Yliruokanen (1980) demonstrated this clearly with the analysis of U with depth in Finnish peat bogs (Figure 2). The concentrations of U were especially high at the bottom of the bogs where it was expected that U-rich groundwaters entered the bog. The mechanisms of retention by organic materials include cation exchange and reduction of U from UO_2^{2+} to UO_2. Leventhal et al. (1986) showed a remarkable correlation between U content and organic-CO_2 content of ancient organic materials (Figure 3). They measured the organic-CO_2 or oxidized carbon content using a pyrolysis technique, measuring the release of CO_2 up to ~390°C. They hypothesized that the organic matter had been oxidized during reduction of the U and, based on this reaction, they showed a nearly stoichiometric relationship between the organic-CO_2 and U contents of these organic materials.

Many Canadian soils are young enough to still contain carbonate minerals near the surface. Complexes of U and CO_3 are relatively stable and can be anionic, resulting in increased mobility of the U. Yamamoto et al. (1973) demonstrated this clearly in a study of the retention of U by soils as a function of the CO_3 concentration in the contact solution (Figure 4). A dramatic shift from 100% sorption to less than 50% sorption onto the soil solids occurred when the CO_3 concentrations in solution exceeded 217 µg CO_3/mL.

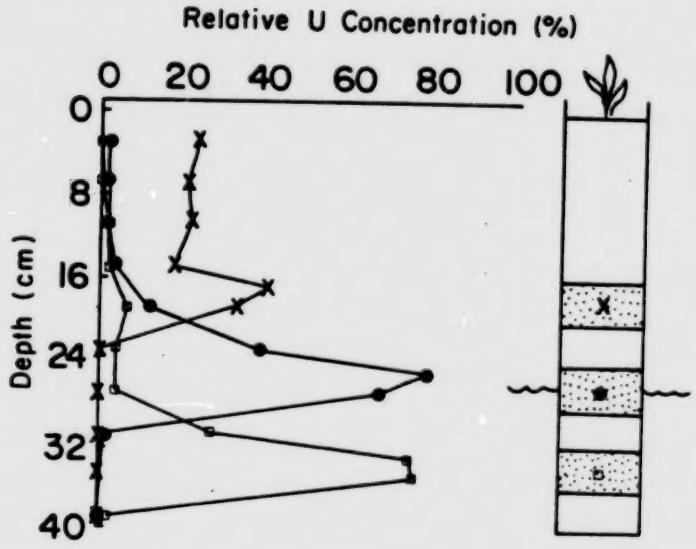


Figure 1 Relative U concentrations (as a percent) with depth in a lysimeter experiment where a two-cm-deep band of U-contaminated soil was placed above, at or below a fixed water table (from Sheppard and Evenden, 1985).

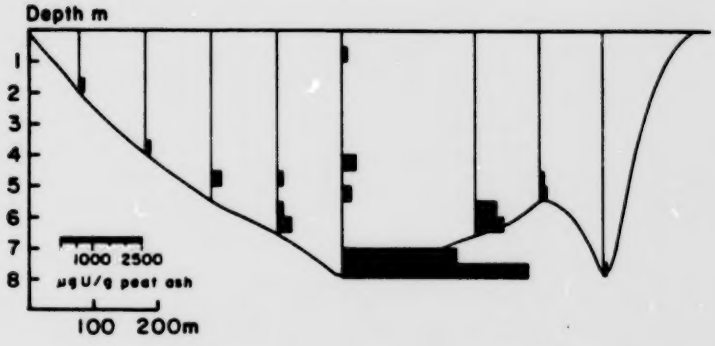


Figure 2 Concentration of U with depth shown on a cross-section of a peat bog where groundwater containing a lov concentration of U enters the bog from rock fissures (from Yliruokanen, 1980).

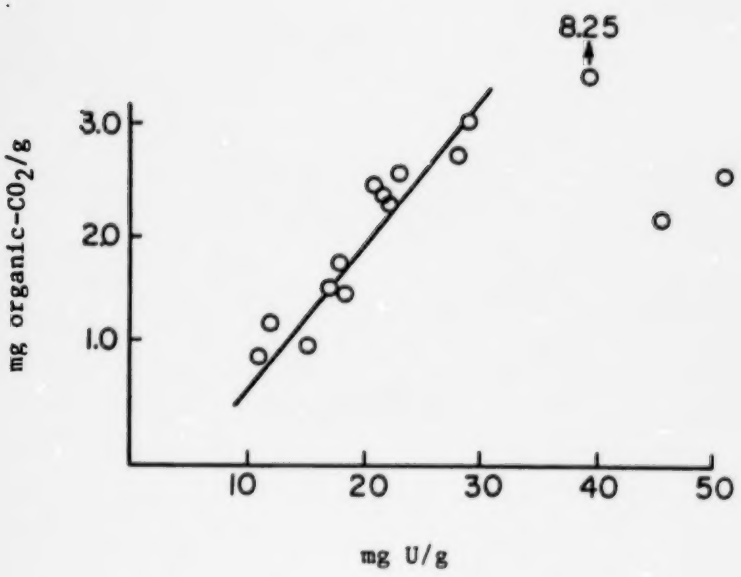


Figure 3 Correlation of the concentrations of organic-CO_2 and U in ancient organic materials, suggesting a mechanistic link between the oxidation of organic carbon and the reduction (and insolubilization) of U (from Leventhal et al., 1986).

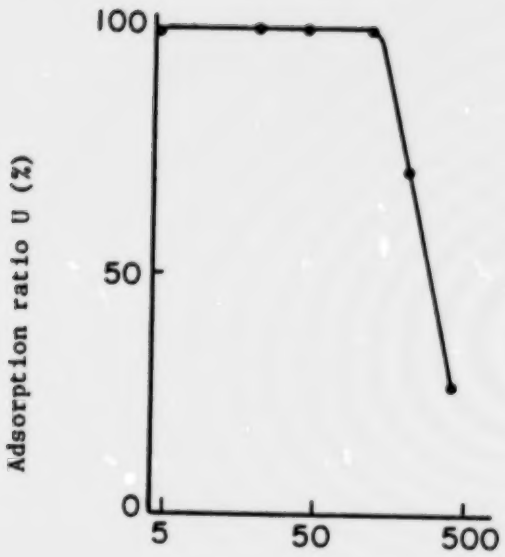


Figure 4 Fraction of U sorbed onto a sandy soil as a function of the concentration of carbonate ion in solution (from Yamamoto et al., 1973).

The interaction of U with mineral soil particles is related to the size of the particles. Smaller particles have a larger surface area per unit mass. The effect of this on the retention of U (and Th and Pb) is shown in Figure 5 from Megumi (1979). He demonstrated that about 10% of the U was held by exchange sites on the soil (extractable in 1 mol/L ammonium acetate), and an additional 40% was associated with iron hydroxide coatings on the particles (extractable in 0.275 mol/L ammonium oxalate). These amounts represent surface-retained U as opposed to U trapped inside the mineral structures. There was a slightly larger proportion of surface-retained U on the smallest particles. These findings reflect the expected, positive relationship between cation exchange capacity and U retention in a soil.

2.1.3 The Behaviour of Th in Soil

Reviews of the behaviour of Th in soils have been given by Sheppard (1980), Harmsen and de Haan (1980), and Adams et al. (1959). Thorium has many characteristics similar to U. Generally, it is less mobile than U, and probably for that reason it has not been as extensively studied. In low pH soils, it may be present as the Th^{4+} ion which is strongly adsorbed by cation exchange sites. At higher pH, the major species in solution will be $Th(OH)_2^{2+}$, although the solubility limit of ThO_2 may be exceeded resulting in virtually no soluble Th. Thus, Th is highly sorbed or otherwise immobilized in soils. Complexes with organic materials can result in uncharged or negatively-charged species which increase the mobility of Th. Similarly, colloids containing Th become important vectors for Th by virtue of the fact that no other mobile forms are common.

Hansen and Huntington (1969) compared Th and U migration based on concentrations found in various horizons of several soil profiles. They hypothesized that soil organic matter tended to immobilize U, whereas it mobilized Th. They found what they described as distinct accumulations of Th in horizons immediately below highly organic horizons. These accumulations were likened to the accumulations of iron (Fe) that occur in lower soil horizons of Podzolic soils as the result of chelation of Fe in the surface horizon by organic ligands.

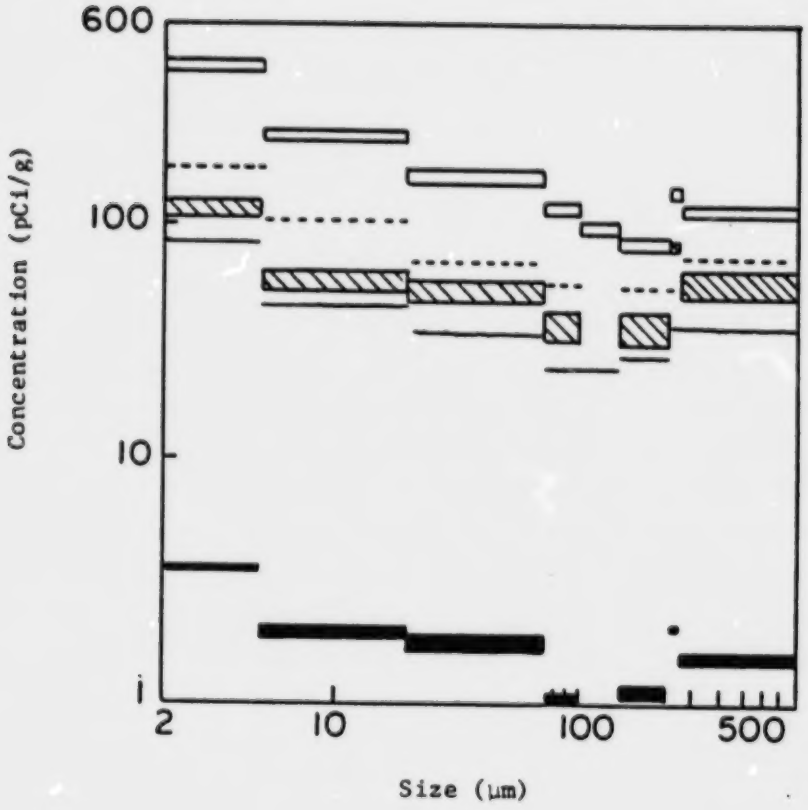


Figure 5 Concentrations of ^{238}U, ^{234}U, ^{235}U, ^{230}Th and ^{210}Pb in various particle size fractions of a soil (from Megumi, 1979).

The concentration differences Hansen and Huntington observed were all less than two-fold.

The retention of Th is related to the particle size (Figure 5) and cation exchange capacity of soils. Megumi (1979) found that even a larger fraction of the total soil Th was associated with particle surfaces than that for U. For Th, about 80% was bound on surfaces with the greatest fraction associated with iron hydroxide coatings. He suggested this may result from a coprecipitation of Th with the iron hydroxide.

2.1.4 The Behaviour of Pb in Soil

Lead is a ubiquitous and highly toxic metal that is a frequent pollutant from industrial and automotive sources. Because of this, much research has been done, usually directed toward understanding the behaviour of Pb from atmospheric or other types of surface deposition. Review papers by Zimdahl and Arvik (1973), and Kabata-Pendias and Pendias (1984) are useful in describing the behaviour of Pb in soils. It is almost exclusively present as Pb^{2+} and several insoluble minerals. Because it is a divalent cation, it is strongly adsorbed by soils and can displace common soil cations such as those of potassium (K^{+}) and calcium (Ca^{2+}) from exchange sites. Complexes of Pb, particularly with organic ligands, may increase its mobility in certain circumstances.

A major difficulty in the study of Pb contamination in food chains is that some native Pb is always present. Kabata-Pendias and Pendias (1984) claim that no soil has been found that is devoid of Pb. Thus, the impact of experimentally-added Pb is difficult to ascertain. McKeague and Wolynetz (1980) surveyed Canadian soils and found concentrations of 5 to 20 µg Pb/g with means ranging from 25 µg Pb/g in the St. Lawrence Lowlands to 15 µg Pb/g in the Interior Plains. There is little doubt that a considerable fraction of the Pb in heavily-populated regions could be due to automobile exhaust. Thus, the finding by McKeague et al. (1979) that Pb content in soils was not correlated with the corresponding clay or organic matter contents is

probably misleading because of the variability in the amount of Pb deposited on the different soils from the atmosphere.

The speciation of Pb in soil solution was evaluated experimentally by Tills and Alloway (1983). Their results (Table 2.2) suggest that in acid soils, most of the Pb was cationic. Some other neutral or anionic forms were also present, presumably the result of organic complexes. In the high-pH soil, which was calcareous, much of the solution-phase Pb was uncharged. However, it must be recognized that only a very small fraction of the total Pb in this soil was in solution.

The influence of organic materials on the amount of Pb in solution was clearly shown by Beveridge and Pickering (1980). Their data (Figure 6) showed that at pH values below 6 or above 7, the solubility of Pb increased substantially, up to 93% of the total Pb present. These findings support observations such as those by Oldfield et al. (1979) that Pb can be relatively mobile in organic soils. Stevenson and Welch (1979) also attributed migration of Pb in a mineral soil to the action of soluble organic complexes. However, despite this effect of organic materials, Pb will remain for many years in soils. This is well illustrated by the data of Livett et al. (1979) which show (Figure 7) a nearly exact correspondence between Pb in layers of a peat soil, where each layer is dated to the time of its formation, and the activities of a nearby Pb smelter. The Pb deposited on the peat by smelter operations prior to 1800 AD are still associated with the peat formed at that time. Thus, the Pb must have been very immobile. Matthews and Potipin (1985) showed a similar long-term retention of fallout ^{210}Pb on mineral soils.

2.1.5 Summary of the Reactions With Soil

The elements U, Th and Pb are all relatively immobile due to adsorption of cationic species and formation of precipitates. Because the simple inorganic species are largely immobile, the migration of U, Th and Pb is dominated by the presence of complex ions and associations with soluble organic materials and colloids. These

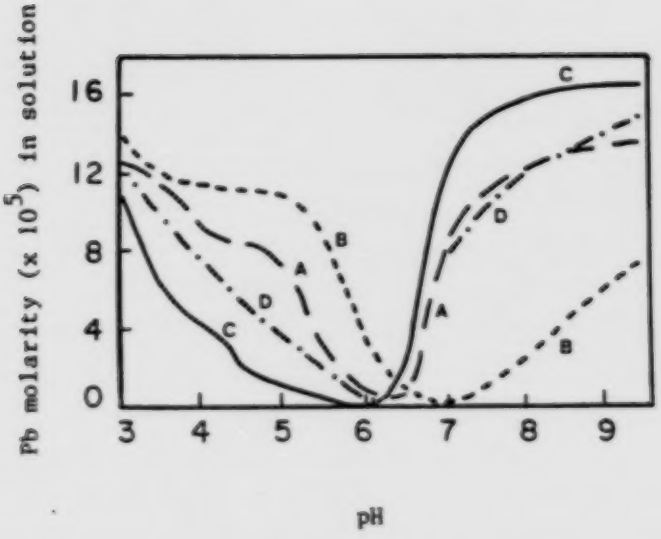


Figure 6 Concentration of Pb in solution as a function of pH and in the presence of several soil organic materials, labelled A, B, C and D (from Beveridge and Pickering, 1980).

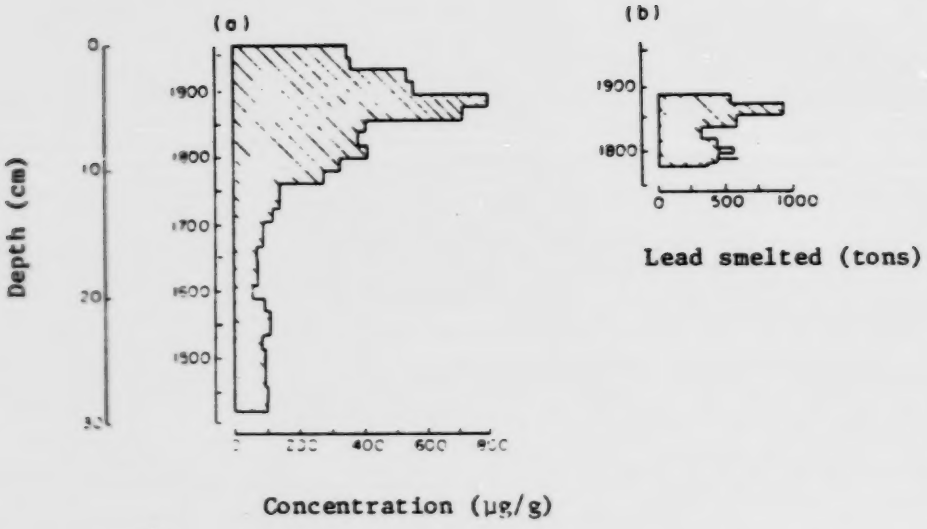


Figure 7 Concentration of Pb with depth in a peat bog (a), where each depth has been dated to its year of formation, and the corresponding activities of a nearby Pb smelter (b) (from Livett et al., 1979).

Table 2.2 The speciation of Pb in soil solution from five heavily-polluted soils (Tills and Alloway, 1983)

soil characteristics				Pb in solution			
total Pb (µg/g)	pH	loss on ignition (%)	cation exchange capacity (me/kg)	concent. (µg/mL)	cationic fraction (%)	anionic fraction (%)	neutral fraction (%)
47000	7.4	8	1.5	0.19	27	2	68
29000	4.1	43	5.6	9.9	97	0	1
3700	4.1	12	2.1	12	99	0	0
990	3.9	44	6.7	2.0	77	5	4
940	5.1	29	5.5	0.090	82	8	0

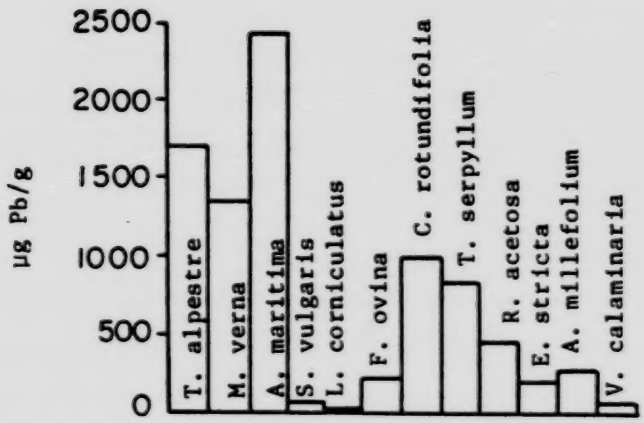
complex forms are often uncharged or negatively-charged, resulting in increased mobility. The relative mobility is U > Pb $\geq$ Th with Pb and Th being effectively immobile in most soil systems.

2.2 Characteristics in Plants

Many of the characteristics of U, Th and Pb in soils, such as chemical speciation and solubility, continue to be important when the elements are absorbed by plants. These characteristics, and others more unique to the physiology of plants, define the mobility and behaviour of U, Th and Pb in plants. This section will review these effects with particular reference to factors important to the interpretation of CR data. An overview of the properties of plants with respect to uptake was given in section 1.2 . A listing of plant species typical of the environs of Canadian nuclear facilities is outside the scope of this report. Harms (1982) provides a taxonomic survey for U tailings in Northern Saskatchewan. This report deals with CR values for native and agricultural plant species typical of any region in Canada. In the following sections, several characteristics in plants that apply to all three elements are discussed, followed by element-specific characteristics.

2.2.1 General Features of Plants that Affect Their Absorption of U, Th and Pb

One of the most obvious features of plants is the diversity of species. Previous research has shown that plant species differ in the degree to which they accumulate U, Th and Pb (see Mahon and Matheves, 1983; Morishima et al., 1976; Sheppard and Thibault, 1983; Nikolaeva, 1978). The results of Baker (1981) for Pb illustrates this point well (Figure 8). Generalizations about uptake by classes of plant types have been reported, but it must be recognized that exceptions will occur. Lower plant types, such as mosses and lichens, tend to have the highest concentrations; but for these species it is not clear what proportion was derived from the substrate, and what proportion came from groundwater and aerial deposition.



Total soil Pb: 20000 µg/g

Figure 8 Species to species variation in the concentration of Pb in plant tissues for plants grown under the same environmental and soil conditions (from Baker, 1981).

Within a plant species, it has also been shown that uptake of U, Th and Pb can differ among cultivated varieties or cultivars. For example, Athalye and Mistry (1972) examined ^{210}Pb uptake by two tobacco cultivars in a solution culture system and found differences in concentration of over two-fold, depending upon the tissue analysed. Although differences among cultivars can be statistically significant, it is not practical to differentiate among cultivars for nuclear assessment purposes.

Several studies report an effect of U, Th or Pb on plant growth and morphology. Several of these seem to reflect chemical- or radiological-toxicity effects, such as stunted growth and mutations (Shacklette, 1964; Nikolaeva, 1978). Others refer to stimulated growth. Care must be taken in interpreting these later results as nitrate salts of U, Th and Pb are especially convenient for experimental systems, yet the plants may respond to the nitrate as a fertilizer nutrient. However, apparently valid cases of stimulated growth due to U (Gulati et al., 1980), Th (Nikolaeva, 1978) and Pb (Khan and Khan, 1983) have been reported. Some authors have implied from data such as these that U, Th or Pb are essential to plant growth in extremely small quantities. Another explanation is that the plants exhibit a stimulatory or hormetic response to low levels of environmental stressors. Sheppard and Regitnig (1987) revieved hormetic responses of plants to low levels of radiation and Stebbing (1982) demonstrated hormetic responses to other stressors, including heavy metals.

A final feature of plants concerns how they interact with the rest of the environment. The very low mobility of U, Th and Pb in soils means that unusual or indirect pathways leading to contamination of plants, such as root intrusion into contaminated groundwaters or deposition of particles on leaves, become potentially important. Dunn (1981 and 1983) and Sheppard and Thibault (1983) found that the concentration of U they observed in aerial plant parts were not correlated to concentrations in the soil, and they concluded that the roots were absorbing U directly from groundwater or from U-enriched soil well below the surface. There is no clear concept of how extensively

plants will absorb radionuclides from selectively-enriched regions in the rooting zone. Drew and Saker (1978) showed that plants can absorb all of the essential nutrients they require from a relatively small proportion of the root system. Similarly, water requirements can probably also be satisfied by a small proportion of the root system. It is likely that the relative uptake of non-essential elements such as U, Th and Pb from various parts of the root system will be proportional to the water absorption by those parts of the root system. Thus, results from studies where plants derived most of their water from contaminated groundwater or some other contaminated zone should be treated with caution. Unfortunately, it is almost impossible in most experimental systems to ascertain whether this condition actually existed. This topic requires further research attention.

The deposition of contaminated particles on aerial plant parts is especially well-documented for Pb. Rabinowitz (1972) reported that up to 90% of the stable Pb associated with plant leaves was deposited there from the atmosphere. Much of the stable Pb in the atmosphere comes from automotive exhausts. However, data from many studies of U, Th and Pb uptake from contaminated areas such as tailings are also suspect. Although it is general practice to wash plant samples in these studies prior to analysis, it is very likely that some of the atmospherically-derived radionuclides are absorbed or otherwise rigidly attached to the plant tissues. The sources of contaminated particles deposited onto leaves include aerosols of local or distant origin, dust of local origin and rain-splashed soil from directly below the plant. Because these mechanisms are common and include materials derived from the soil directly below the plant, it is often impossible to differentiate the relative contributions of root uptake and atmospheric deposition in the studies reported in the literature.

2.2.2 Behaviour of U in Plants

The reviews by Sheppard (1980) and Cannon (1952) discuss the key characteristics of U in plants. Dunn et al. (1985) have completed an extensive review of U concentrations in plants. In this section, we

will mention the key characteristics and illustrate them with data from specific references.

Uranium is strongly sorbed to cell wall materials. However, once it enters the symplast, it is moderately mobile and moves throughout the plant. It can be remobilized from older tissues and moved toward never tissues. In this respect, it is an analogue of phosphorus (P), and the concentrations of U and P in plants have been shown to be correlated.

The sorption of U to cell wall materials was illustrated by Andreyev et al. (1962). They examined U sorption onto plant derivatives such as cotton, wood flour, lignin and cellulose preparations. Sorption was pH-dependent ranging from 35 to 90% of the U in solution, and was approximately linear with the concentration of U in solution for most of the plant materials. They hypothesized that ion exchange, specific chemisorption and physical adsorption (probably meaning occlusion) were responsible. Robards and Robb (1972) and Weinberger and Murthy (1985) showed similar effects on cell walls of living plant roots. A major consequence of this sorption to cell wall materials is that data of U concentration in plant roots, for example for edible-root crops, is strongly dependent on whether or not the root surface is removed prior to analysis. Since the U on the root cell walls may be almost irreversibly sorbed, washing of roots in mild solvents is not sufficient to extract this non-biologically retained U.

The distribution of U in plants and the change in concentration with time are related phenomenon since old tissues are usually lower on the plant morphological structure. Garten (1980) showed progressively higher concentrations of U in tree leaves as the growth season progressed, and Mahon and Matheves (1983) observed a similar trend for one of five plant species. Conversely, Dunn (1983), Cannon (1952) and Sheppard and Thibault (1984) showed no consistent trends with time in a given season. However, Dunn (1981) did show a trend with age of perennial tissues (Figure 9). The highest U concentrations were found in two- to three-year-old twigs, with lower concentrations in new growth and older wood. Gruzdev and Rubtsov (1972) reported similar

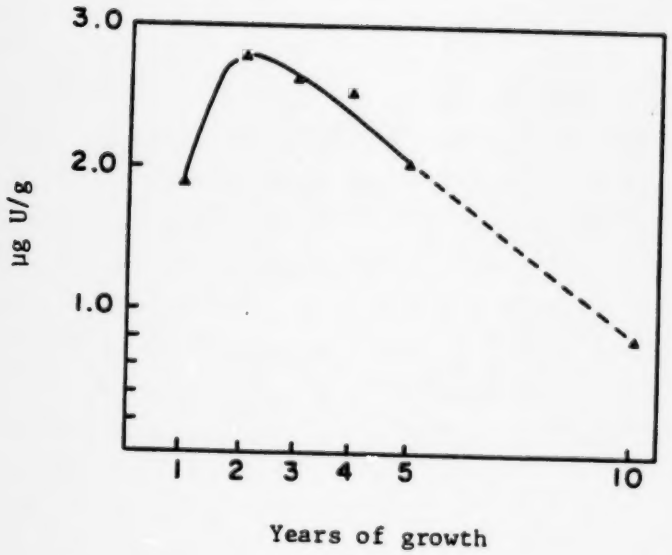


Figure 9 Concentration of U in spruce twigs as a function of the age of the twig (from Dunn, 1981).

trends. We explain these results by hypothesizing that accumulation of U by active tissues is a relatively slow process. Thus, rapidly growing new tissues have relatively low U concentrations. The U concentration reaches a maximum after rapid growth stops. An eventual decrease reflects a slow loss of U from older, less active tissues. Thus, the reported effects or lack of effects of time and tissue age reflect the balance of plant growth rate and U accumulation rate, coupled with the loss of U as tissues become less active.

The analogy of U to P is worth noting. The fact that U is present in elevated concentrations in most P-bearing rocks used for fertilizer production is a demonstration of the analogy. The relevance of the analogy to U uptake is not straightforward. Morishima et al. (1977) showed a relatively strong correlation of U and P concentrations in leafy vegetables grown on farm fields (Figure 10), suggestive of an analogous behaviour in plant uptake at low concentrations of P and U. However, if the analogy holds, a negative correlation might be expected if U or P concentrations were especially high.

An interesting, positive relationship between U concentrations and the ash content of plants and plant parts was noted by Dunn (1981) and Morishima et al. (1976). An explanation may be that U behaves similarly in the plant to P, Ca, magnesium and K, the major constituents of plant ash. Much of the data on U concentrations in native plant species is based on ash weight, since this is the norm in biogeoprospecting. It may be that expressing all plant U concentrations on an ash-weight basis would be a more consistent practice than using a dry-weight basis. This should be investigated further as it may further reduce the variability of CR values.

2.2.3 Behaviour of Th in Plants

The behaviour of Th in plants has not been extensively studied and few generalizations can be made (Sheppard, 1980). It has many characteristics similar to U, especially in terms of which tissues contain the highest concentrations (Nikolaeva, 1978; Gruzdev and Rubtsov, 1972). No other reliable analogue elements have been identified.

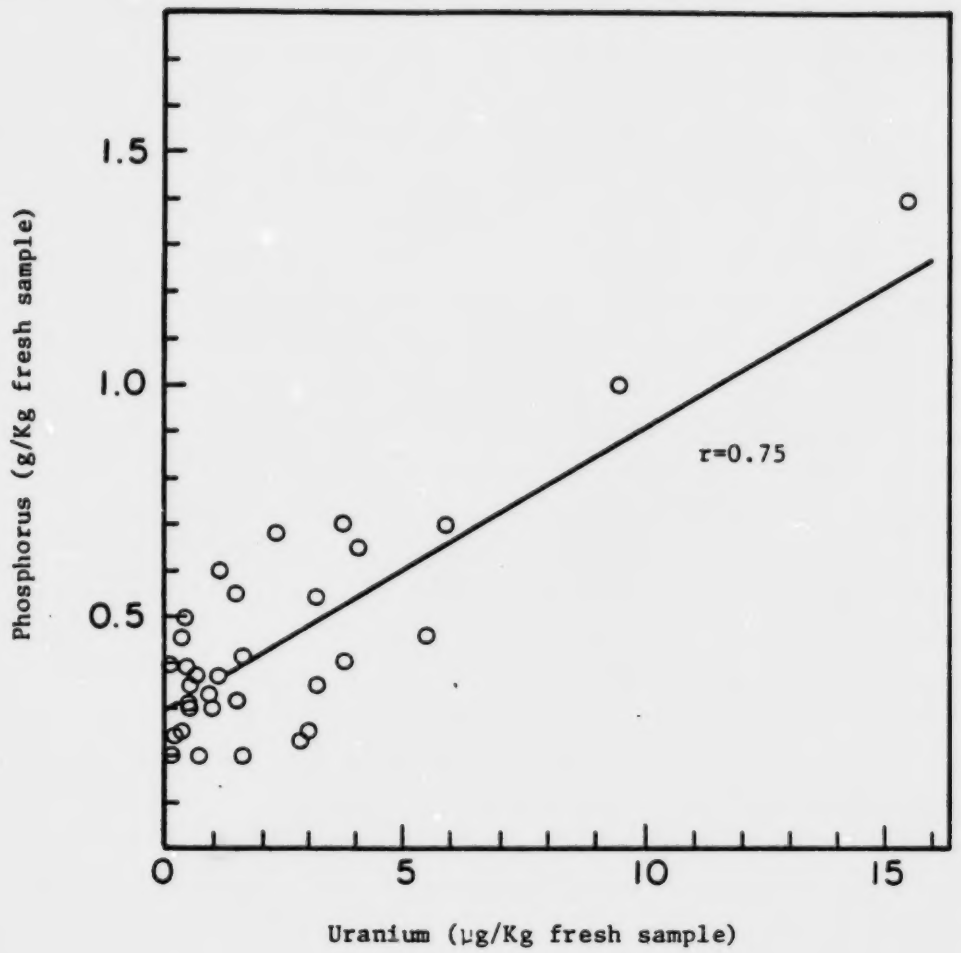


Figure 10 Correlation between U and P concentrations in leafy vegetables grown on farm fields where r is the simple correlation coefficient (from Morishima et al., 1977).

2.2.4 Behaviour of Pb in Plants

Concise reviews of the behaviour of Pb in plants are given by Mengel and Kirkby (1979) and Kabata-Pendias and Pendias (1984). Briefly, the Pb in plants is accumulated in cell walls, much like Ca, and this tends to limit the mobility and toxicity of Pb. Uptake of Pb is probably passive and concentrations in the shoot are usually less than 3% of that in the roots. Thus, little of the Pb absorbed by roots is transferred to the shoot. Much of the Pb in leaves is derived from the atmosphere. Although some of this atmospherically-derived Pb can be washed off, there is no doubt some is absorbed or otherwise chemically bonded to the leaf tissues. The mechanism of Pb toxicity is not clear, but trialkyl Pb may be formed in plants and this is a powerful mutagen causing derangement of the spindle fiber mechanism during cell division. Concentrations as high as 2700 µg Pb/g have been observed in plants in Canada (Kabata-Pendias and Pendias, 1984). Plant species vary markedly in the degree to which they accumulate Pb, as shown by data from Baker (1981) in Figure 8. Plant roots (especially relevant for those root crops eaten with the skins or root surfaces intact) will have the highest concentrations of soil-derived Pb within the plant. However, the deposition of Pb onto leaves by aerosol, dust, and rain-splashed pathways can overshadow the root-uptake pathway, and frequently confound the data in the literature. Leavitt et al. (1979), Athalye and Mistry (1972) and Mahon and Matheves (1983) all generally agree that older tissues have higher concentrations of Pb.

An analogy is sometimes drawn between Ca and Pb. Some of the toxicity of Pb may be the result of the displacement of Ca by Pb from critical biochemical pathways. The most direct evidence is the fairly general finding that higher levels of Ca, for example after liming of a soil, result in reduced uptake of Pb.

2.2.5 Summary of Characteristics in Plants

The elements U, Th and Pb are not absorbed to a great extent by plants. They tend to be immobilized by adsorption onto cell wall

materials. This inhibits absorption into the plant and slows redistribution throughout the plant once absorbed. The relative mobility in plants is U > Pb ≥ Th. Because Th and Pb in particular are not very effectively absorbed from the soil by plant roots, the contamination of aerial plant parts by the deposition of contaminated particles becomes an important pathway. It is often difficult to differentiate the relative contributions of the root-uptake and deposition pathways to the Th and Pb found in aerial plant parts.

Effects of plant species, tissue and age on concentration in the plants have been documented. However, generalizations are not reliable, probably because these effects are dependent upon more-basic but seldom-documented variables such as growth rate, evapotranspiration rate and radionuclide uptake rate.

2.3 Dependence of Uptake by Plants on Behaviour of the Radionuclides in Soil

The discussions in sections 2.1 and 2.2 dealt independently with the behaviour of U, Th and Pb in soils and plants, respectively. Clearly, if concentrations in plants vary with age, as an example, then the corresponding CR values will also vary with age. The connection with CR values is less direct for factors specific to soils. It is reasonable to expect that if a soil factor affects the behaviour of U, Th or Pb in the soil, it will also affect plant uptake and the corresponding CR values. This section discusses evidence to support this relationship.

The importance of the soil in controlling CR values is especially evident in studies where plant concentrations were shown not to be correlated with total concentrations in the soil. Lack of such correlation for U was reported by Amiro and Sheppard (1987), Dunn (1981 and 1983), and Sheppard and Sheppard (1985), for Th by Evans and Eriksson (1983), and for Pb by Bates et al. (1975). Although some authors cite these findings as evidence that the CR model is not valid, we view them as evidence that soil factors are important. Our hypothesis for these data is that the variability in plant uptake

caused by other soil factors overshadoved the variability caused by the range in total concentration in the soil. Experiments where other soil factors were less variable or were controlled usually resulted in a significant correlation between plant and total soil concentrations (Sheppard and Evenden, 198x; Mahon and Matheves, 1983; Mordberg et al., 1976).

The lack of a consistent correlation between plant concentrations and total concentrations in soil has long been recognized in agronomy. The concept is simply that plant roots have physical and chemical access to only a relatively small fraction of the total amount of an element in the soil. In order to predict fertilizer requirements for crops, agronomists have developed methods to measure the plant-available fraction (for example; Sheppard and Bates, 1982). This fraction is usually measured using an extractant that removes the water-soluble amount of the element in the soil, plus some less-easily extracted fractions. The relationship between this concentration of plant-available element and the actual plant concentrations is again not exact, but it is less variable and less dependent on soil conditions than the relationship between the total concentration in the soil and plant concentrations.

This concept has been applied to radionuclide CR data by Baes (1982) and Sheppard (1985). They both related CR values to the solid/liquid partition coefficient (K_d) in soils. In effect, this relationship implies that plant concentrations are dependant upon the concentrations in soil solution. Sheppard and Evenden (198x) examined this relationship for U and Pb in a study where the total soil concentrations of each element ranged from background to 10,000 µg/g. The concentrations of Pb were well correlated with both the total and water-soluble fractions of Pb in the soil. This was not the case for U. The plant concentrations of U increased approximately in proportion to total soil concentrations up to about 1000 µg U/g. Above this, plant U concentrations increased much more rapidly with increases in total soil concentration. However, the concentration of U in the soil solution followed a similar trend. The net result was that plant concentrations were linearly related to the soil solution U

concentrations throughout the range of soils. This finding suggests that any soil factor that varies the amount of U in the soil solution will also vary the uptake of U, and hence the CR.

The relationship between concentrations in plants and in soil solution does not always hold. Gerritse et al. (1983) found a negative correlation for Pb, and Sharma and Kalin (1982) also concluded that for Pb, the relationship was not always reliable. Miller et al. (1975) showed an interesting and possibly related concept for Pb (Figure 11). They measured the maximum sorption capacity for Pb of several soils which they had used in an experiment. The experiment had involved the growth of plants on these soils with the addition of several levels of Pb. The CR values for Pb varied with the different soils. However, when they related the Pb concentration in the plants to the ratio (added Pb)/(maximum Pb sorption capacity), they found the differences between the soils were effectively eliminated. The implication is that plant uptake of Pb was related to the Pb sorption capacity of the soils. This finding may support the conclusion of Sharma and Kalin (1982) that Pb uptake by plants is dependent upon a threshold soil Pb content.

In summary, the relationship of soil factors to CR is not direct. The dominant effect of soils on CR values seems to be mediated by effects on the solubility or mobility of the elements in the soil. Although accounting for effects on mobility may not explain all of the influence of soil factors, it is likely the single most important effect.

2.4 Overview of Key Factors Affecting CR Values

The factors affecting CR values can be broadly categorized as either related to soils or to plants. The soil factors can influence CR values a number of ways, but the dominant effect seems to be on the mobility of the elements in the soil. Factors that increase mobility in the soil probably also increase uptake by plants. The review indicated that mobility in soil could vary with cation exchange capacity, pH, organic matter content, carbonate content and

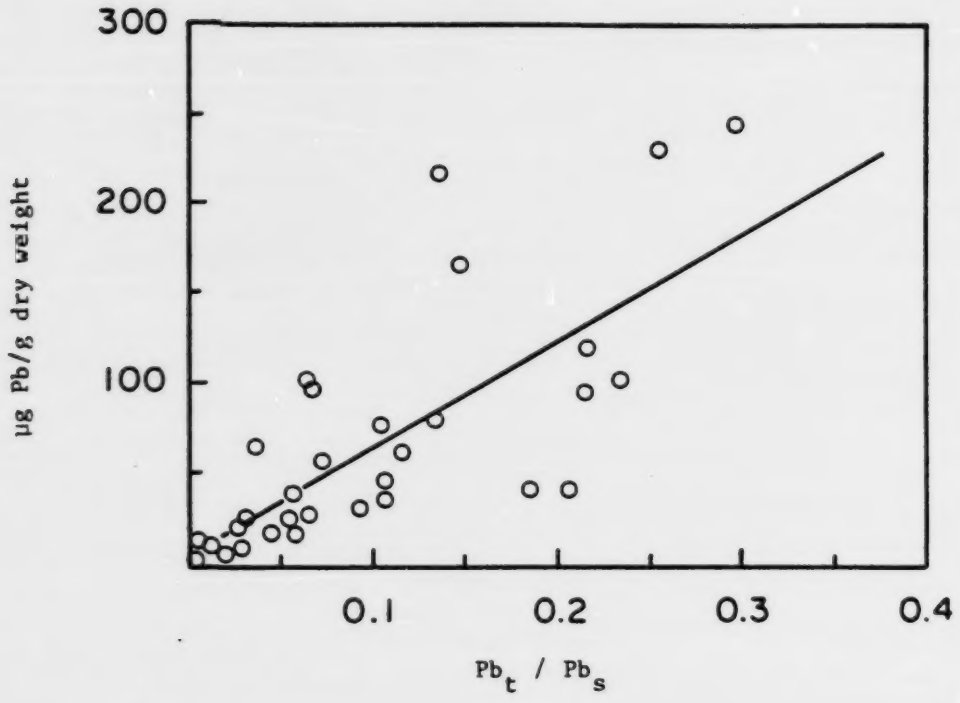


Figure 11 Concentration of Pb in corn plants as a function of the ratio of the amount of Pb added to the soil (Pb_t) to the maximum Pb-sorption capacity of the soil (Pb_s) (from Miller et al., 1975).

oxidation/reduction state. The relationships of these factors to each other and to other commonly reported soil factors is given in Table 2.3 . We suggest that from the viewpoint of plant uptake, the order we have listed them in corresponds to their decreasing order of relative importance, with cation exchange capacity as the most important factor.

Plant factors can also influence CR values a number of ways. There are probably two basic underlying factors, and these factors are interrelated. One is the effect of plant species with its implied effects of growth form, the part of plant used in the food chain and the propensity to accumulate non-essential elements. The second is the effect of growth rate which reflects culture conditions, temperature, and nutrient and moisture supplies. Clearly, growth rate is also a function of both species and soil conditions.

3. COMPILATION OF CR VALUES FROM THE LITERATURE

3.1 Tabulated Compendium of Data

The tables in this section summarize all of the CR data we were able to derive from the literature. The references we used were identified in searches of computer-indexed literature, and by citation from recent papers. Some valuable references were undoubtedly missed, especially for Pb, where much of the published information is indexed with reference to non-nuclear applications. It should be noted that the CR concept is not broadly used outside the nuclear industry.

We computed CR values for those references where the basic concentration data were presented, but the CR values were not reported. The assumptions we made to do this, and the detailed sources of all of the data, are itemized in Appendix A. We computed average CR values, as geometric means (GM), among any groups of values that the authors found not to differ significantly, or that we concluded did not differ substantially. The comments in the tables reflect characteristics of the original studies that might bias the

Table 2.3 Relationship of key soil factors to other commonly-reported soil characteristics.

Key Factors	Related Soil Characteristics[1]	Related Soil Types[2]
cation exchange capacity	texture parent material pH organic matter content	Podzolic Organic
pH	parent material developmental age of soil organic matter content	Podzolic
organic matter content	parent material depth to water table	Organic Podzolic
carbonate content	parent material developmental age of soil pH	Luvisolic
oxidation/reduction status	depth to water table texture biological oxygen demand	Gleysolic

[1] "Related soil characteristics" refers to commonly reported factors that are usually closely correlated with the corresponding key factors.

[2] "Related soil types" refers to soil types that are characterized by extremes of the key factors.

reported CR values. The units for the concentrations of the elements, given as ancillary information in Tables 3.1 to 3.3, are those presented by the original authors.

Table 3.1 Compilation of CR values for U

reference	type of study	soil information	plant information	number of sites or observations
Adams et al. (1974)	greenhouse 2.3-L pots	"meadov soil" 5% organic 4-90x10^{4}dis/min g^{232}U/g as nitrate	alfalfa	3
			barley plant	3
			barley grain	1
Amiro and Sheppard (1987)	field sampling at tailings area	37-379µg U/g	cattail pollen	21
			cattail upper vegetative	29
			cattail rhyomes	29
Arkhipov et al. (1985)	replicated in field, 15-m^2 plots	medium to heavy texture chernozem pH 6-7, 400 me bases/kg, 6% organic, 20-93 Bq ^{238}U/kg	wheat grain	4 reps x 2 yrs x 5 treatments
Baeva et al. (1975)			several species	
Bondietti et al. (1979)	field, contaminated for over 30 yrs	floodplain soil	leaves and stems	15
			fruit, vegetable or seed	15
CBCL Limited (1985)	field sampling of natural high-background bog	peat deposit adjacent to mineralized talus, 100-28000 µg U/g	trees, shrubs lickens and moss	very fev samples had detectable U
Cannon (1952)	field sampling of mineralized areas	see Comments	15 native dryland species	11
		soils or unconsolidated rock materials containing roots	tops of several native species	27
			roots of above	18
			tops and roots of above	36

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
		0.005	Arithmetic means presented by author
		0.0017	
		0.00014	
	0.001 (-7.1±0.9)		
	0.003 (-5.7±0.9)		
	0.6 (-0.5±0.7)		Washed in water
	0.0076 (-4.9±0.4)		We presumed plant dry weight was presented
	0.035-0.186		
	0.018 (0.009-0.02)		Plants included soybeans, snap-beans, millet, squash, tomatoes carrots and radish
	0.00018 (0.0005-0.03)		
	0.00001-0.0001		The authors suggested these very low CR values were related to the fact that the substrate was peat.
		0.17 (-1.7±2.6)	Not directly relevant because parent rock was defined as the substrate
		1.04 (0.04±0.9)	Data selected for sites where soil-like substrate was described
		0.9 (-0.1±2.2)	
	0.028 (-3.6±2.5)		Computed from above using individual measurements of plant ash content

Table 3.1 Continued

reference	type of study	soil information	plant information	number of sites or observations
Davy and Conway (1974)	field sampling of mineralized area	clay-bottom seasonally-flooded creeks, 0.36-1.5 pCi U/g	pasture grasses	5
			palin fruit	1 mean given
		1.5-7.2 pCi U/g, 5% loss on ignition	paperbark leaves	5
Dreesen and Marple (1979)	small scale bioassay technique using plants	tailings soil overlain by each of 3 soils 70-90 µg U/g overall	grass	3 soils(4-6 internal replicates)
			shrub	3 (see abvove)
		sand, loam or clay soil, 1.5-3.6 µg U/g	grass	3 (see above)
			shrub	3 (see above)
Dreesen et al. (1982)	greenhouse study	soil-covered tailings 60-280 µg U/g 0.56 to 45% leachable	grass	not given
			shrub	not given
		surface soil	grass	not given
		2.5 µg U/g	shrub	not given
	field sampling around a U mill	contaminated soils 37-73 µg U/g	annuals, grasses and legumes	4 means given
		field, garden and "background" sites 3.1-9.0 µg U/g	as above	7 pairs of high and lov values given
Dunn (1981)	field sampling of U anomaly	podzol and peat soils 0.8-24 µg U/g, weighted mean 4.7 µg U/g	spruce twigs	22
			spruce needles	22
			conifer trunks	132
			shrub leaves	138
			shrub stems	138

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.029 (-3.5±1.6)		Some of the U content of the plants may have been ad/absorbed directly from flood waters
0.01		0.34	see above
	0.007 (-4.9±0.5)		see above
	0.002 (-6.2±0.2)		The technique employed 2.6-L pots with 800g of tailings overlain by 880 to 1320 g of sand, loam or clay soil
	0.02 (-3.9±0.7)		
	0.027 (-3.6±0.2)		These data from 2.6-L pots of each soil, no tailings material
	0.01 (-4.3±0.7)		
	0.00089		Plants were grown for 6 months, various tailings materials were notably different in the leachability of the U they contained.
	0.010		
	0.028		
	0.016		
	0.10-0.54		Plants may have been exposed directly to contaminated groundwaters on these sites.
	0.034(-3.39±0.91) (0.0049-0.28)		
	0.25	18(11-33)	The authors found no correlation between plant and soil U concentrations and felt that the plants may have absorbed U from rock fissures that interacted with the underground U anomoly
	0.009	3 (2-5)	
	0.0004	0.12 (<.1-2)	
	0.24	7.4 (4-18)	
	0.17	12 (7.6-21)	

Table 3.1 Continued

reference	type of study	soil information	plant information	number of sites or observations
Dunn (1983)	field sampling of U anomaly	podzol soils in boreal environment 1.3-3.2 µg U/g	black spruce twigs	34
Evans and Eriksson (1983)	field sampling of agricultural area mineralized with U	37 sites ranging in pH, clay and organic matter contents, 70±57 Bq ^{238}U/kg	mature rape, barley and wheat grain	8 fields, 2 isotopes
			as above, straw	as above
			as above, mid-season	as above
Frindik (1985)	field sampling	several sites	wheat, barley and rye	not given
			10 vegetables	not given
			4 fruits	not given
Garten (1980)	field sampling	clay floodplain soil pH 6.7-7.5, 3000-4000 fCi ^{238}U ^{234}U/g	boxelder tree leaves	3 trees
Garten (1981)	field sampling	shoreline sediment 4000-58,000 fCi/g	Eleocharis (wetland plant)	14
Garten et al. (1981)	field sampling	clay floodplain soil pH 6.7-7.5, 10 µg U/g	fescue	9 plant samples
Gilbert et al. (1976) from Garten (1978)	field measurement	arid region		
Gulati et al. (1980)	greenhouse study	1.5-6 µg U/g	mature wheat	GM of 9 means
			mature tomato	GM of 9 means

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
	0.94	66 (4.2±1.6)	
	0.0010 (-6.9±0.77)		
	0.0010 (-6.9±0.84)		
	0.00075 (-7.2±0.55)		
0.0011 (0.0005-0.0018)			
0.0010 (-6.89±0.53)			
0.0005 (-7.6±1.4)			
	0.0007-0.013		
	0.0044		Soil and plant concentrations ranged 2-fold above and below the means
	0.01		
	0.001-0.1		Garten thought data from these authors included dust loading
	0.029		CR values not consistently dependent on soil U concentrations but did increase with increases in irrigation water (plant yields did not increase with increases in irrigation water)
	0.043		

Table 3.1 Continued

reference	type of study	soil information	plant information	number of sites or observations
Hanson and Mieva (1976) from Garten (1978)	field measurement	dry land		
Lopatkina et al. (1970)	field sampling	marshy flood plain soil 1-20 µg U/g	leaves and twigs of birch, alder, willow, etc.	GM of 37 values
		20-100 µg U/g		GM of 14 values
		100-2000 µg U/g		GM of 31 values
Mahon and Matheves (1983a)	field sampling	till and alluvium pH 5.2-6.6, up to 125 µg U/g (7.0± 7.5 µg U/g)	native forbs, grasses, shrubs, and trees	weighted GM of 8 means, each mean of 2-10 observations
Mahon and Matheves (1983b)	field sampling	lake sediment 14±5 µg U/g	pond-lily	weighted GM of 2 means, each mean of 5 or 7 observations
Milosevic et al. (1980)	field sampling of uncultivated pasture	podzolic soils 0.5-1.8 µg U/g	pasture grasses	8 sites
Moffet and Tellier (1977)	field sampling of plots	revegetated U mine tailings, sands and fines, 16±3 µg U/g	4 grass species grown for 4 years	5 replicate plots per species/site GM of 8 ratios
Mordberg et al. (1976)	field sampling	chernozem 4-5% organic, light clay, 2.5 mg U/kg	wheat grain	not given
			corn grain	not given
			buckwheat grain	not given
			dried peas	not given
			millet grain	not given
			sunflower seeds	not given

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.02-0.34		Authors thought dust loading on vegetation contributed to plant U concentrations
		0.056 (-2.9±1.27)	Authors believed that CR values decreased as U concentration in soil increased
		0.0071 (-5.0±0.68)	
		0.0021 (-6.2±1.3)	
	0.0076 (-4.9±0.58)		Authors computed separate CR values for 8 species, we averaged these together
	0.015 (-4.2±1.2)		Lake water contained 20-30 mg U/L
	0.0030(-5.8±1.3)		We assume the concentrations were on a dry weight basis
	0.0017 (-6.4±0.46)		4 species and 5 sites did not differ significantly based on statistical tests of soil and plant concentrations.
0.0038±0.0025			Plant (grain and seed) concentrations expressed on basis of "commercial" moisture content which was probably 7 to 16% moisture by weight. Means were computed by the authors and apparently are arithmetic, the measure of variance is either the 90% confidence level or the range.
0.0017±0.0003			
0.0044±0.0017			
0.0001±0.003			
0.0009±0.0016			
0.0027±0.0006			

Table 3.1 Continued

reference	type of study	soil information	plant information	number of sites or observations
Morishima et al. (1977)	field sampling	2.9-5.6 µg U/g	leafy vegetables	21 plant samples
			root vegetables	8 plant samples
			berries (including tomatoes, egg plant, pumpkin, fruit, etc)	17 plant samples
			wheat and rice grain	4 plant samples
			potatoes	17 plant samples
Morishima et al. (1976)	lysimeters each 62.5 L	sandy soil, pH 4.8, 8% loss on ignition, soil limed to pH 7.0 and well fertilized, 1000 µg U/g above background	radish leaves and roots	not clear
			cucumber leaves and fruits	not clear
			pimento leaves and fruits	not clear
			radish leaves and roots when deficient in P	8 plants
Prister and Prister (1970)	pot (hydroponic?) culture	quartz sand 7 kg/pot	beans	not given
Rayno et al. (1980)	field sampling	area around U mine and tailings ponds, arid region 3-18 pCi ^{238}U/g	4 forage species	not given
Rumble and Bjugstad (1986)	field sampling	tailings covered with 10-30 cm of soil, pH 7-8, 13 µg U/g in top 20 cm	2 grasses and 1 forb	30 sets
		control area 3-9 µg U/g	as above	not given

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
0.00073		0.036	Irrigation water containing 0.016 µg U/L was used; the method, amount or frequency of use was not specified, ash contents inferred from data presented by authors for five plant types were 2%, 1.3%, 0.8%, 1.3% and 1.1%, respectively.
0.00018		0.014	
0.00017		0.021	
0.00015		0.012	
0.000085		0.0077	
		0.022	
		0.029	
		0.042	
		0.17	
	0.009-0.18		The methodology is not clear, it seems to be a hydroponic sand culture system using relatively inert quartz sand as the rooting medium.
	0.01-0.11		The proximity to uncovered tailings and the arid climate may have facilitated contamination by dust, although authors did wash samples
	0.0075-0.025		Authors did not compute means as geometric means, no discussion of plant roots entering the tailings materials, we assumed the CR values were on a dry weight basis.
	0.0048-0.0077		

Table 3.1 Continued

reference	type of study	soil information	plant information	number of sites or observations
Schreckhise and Cline (1978)	small scale growth cabinet 4.3 kg soil/pot	silt loam 6.8 nCi ^{232}U/g	barley grain	not given
			barley leaf and stem	not given
			pea seed	not given
			pea stem and pod	not given
			pea leaf	not given
Sheard et al. (1986)	field sampling	boreal zone, regosolic soils, 0.9-2 µg U/g (Wollaston Lake area)	4 tree species	12
			5 shrub species	27
		0.4-0.5 µg U/g (Cluff Lake area)	3 tree species	10
			4 shrub species	15
		0.9-325 µg U/g (Uranium City area)	4 tree species	18
			7 shrub species	19
Sheppard et al. (1982)	pot culture 1 L/pot	mixtures of waste site soil and a sand soil	pine and tomato seedlings	15 pot/species
Sheppard and Thibault (1983)	field sampling	boreal mineral and organic soils 0.01-2 µg U/g ashed soil	stems of 2 conifer species	133 samples
			needles of 2 conifer species	109 samples
			twigs and leaves of birch	15 samples
			twigs and leaves of 3 shrub species	123 samples

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
	0.00016		
	0.0036		
	0.00054		
	0.0033		
	0.017		
	0.029 (-3.6±1.7)		Needle and twig samples collected as appropriate for specific food chains.
	0.050 (-3.0±1.2)		
	0.049 (-3.0±1.8)		
	0.13 (-2.0±2.0)		
	0.0016 (-6.4±2.0)		
	0.0026 (-6.0±1.3)		
	<0.07		
	0.0038	0.15(-1.9±1.3) (0.008-2.5)	
	0.016	0.051(-3.0±1.2) (0.009-1.1)	
	0.004	0.11(-2.2±0.94) (0.021-1.0)	
	0.002	0.052(-3.0±1.5) (0.002-2.1)	

Table 3.1 Continued

reference	type of study	soil information	plant information	number of sites or observations
Sheppard et al. (1984)	outdoor lysimeters 8 L/lysimeter	Orthic Regosolic sand, pH 8.6 2-15 µ U/g in top 15 cm	alfalfa and Swiss chard	36 lysimeters
Sheppard and Evenden (1985)	outdoor lysimeters 8 L/lysimeter	Orthic Black Chernozem clay loam, pH 8.1 13-197 µg U/g	barley plants at maturity	20 lysimeters
		107-175 µg U/g	as above, root development restricted by high water table	4 lysimeters
Sheppard et al. (1985)	outdoor pot culture 2 L/pot	a waste site soil (3-43 µg U/g) and a clay loam soil (artifically contaminated to 10-100 µg U/g)	two-year-old Scots pine seedlings	83 plants
Smith et al. (1982)	replicated plots (10x10 m) downwind of U separations facility	600, 26 and 500 fCi/g of ^{234}U, ^{235}U and ^{238}U	winterwheat straw	10 plots
			winterwheat grain	5 plots
Titaeva et al. (1979)	field sampling	soddy meadov, 1.3 µCi ^{238}U/g ash	grasses and forbs	not given
		19 µCi ^{238}U/g ash gleyed podzol,	grasses and forbs	not given
		62 µCi ^{238}U/g ash	tree species	not given
Tracy et al. (1983)	sampling of contaminated gardens (Port Hope)	7.5-420 µg U/g	leafy vegetables	3 samples
			roots and stems	6 samples
			fruit	8 samples

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.26-1.4		The U was placed in discrete layers 15 or 30 cm below the soil surface, roots penetrated to 45-60 cm. The U migrated upwards to varying degrees. Plant growth was quite poor in this soil.
	0.033(-3.4±0.57)		The experimental design required that the U migrate into the root zone to become accessib to roots. This occurred as the plant grew. CR values based on soil U concentrations at the end of the experiment.
	0.008(-4.8±0.71)		
	0.03	0.0014	
	0.008		Deposition rates of 106, 4 and 110 fCi/m^2·day ^{234}U, ^{235}U and ^{238}U during the study, CR values were not different among the isotopes, coefficients of variability in plant concentrations were about 50 and 170% for straw and grain, respectively
	0.0006		
		0.12(-2.1±1.9)	The authors alluded to but did not fully explain what may have been an anthopogenic origin of the radionuclides.
		0.055(-2.9±1.7)	
		0.017(-4.1±1.1)	
0.00015(-8.8±1.2)			
0.00027(-8.2±1.3)			
0.00008(-9.4±1.8)			

Table 3.1 Continued

reference	type of study	soil information	plant information	number of sites or observations
Van Netten and Morley (1981)	pot culture in growth chamber	five soils, pH 3.6 to 7.7, 3-570 µg U/g	oats, above ground plant	not given
Van Netten and Morley (1982)	pot culture in growth chamber	six soils, pH 5.6 to 7.8, 2.0-560 µg U/g	barley stalks and grain	not given
Van Netten and Morley (1983)	pot culture study in greenhouse	as above	radish roots radish stalks	not given
Walker (1979)	field sampling of high-background area	sandy, possibly podzolic 0.6-14 µg U/g ash	pine, spruce, blueberry and labrador tea (leaves, needles and twigs)	3-34 samples of each vegetation type and part
Whicker and Ibrahim (1983)	field sampling	natural background and reclaimed areas 1-4 pCi/g	grasses, forbs and sagebrush	20 samples
Yamamoto et al. (1968)	sampling of farm fields	volcanic ash soil, 1.9-4.0 µg U/g	several leafy vegetables	2 sites x 3 years x 3 replicate samples

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
	0.068(-2.7±1.2)		
	0.0055(-5.2±1.9)		
	0.040(-3.2±1.6)		We assumed the CR values
	0.0066(-5.1±1.9)		given were on a dry weight basis, we average results of 6 soils
		0.72(-0.33±0.50)	
	0.00016-0.017		
	0.12(-2.2±0.83)		Some U in plants may have been derived directly from irrigation water

Table 3.2 Compilation of CR values for Th

reference	type of study	soil information	plant information	number of sites or observations
Arkhipov et al. (1985)	see U	see U 32-59 Bq ^{232}Th/kg	wheat grain	4 reps. x 2 years x 5 treatments
Ashkinazi (1982)	field sampling, ^{238}Th from fertilizer materials	medium loam, podzolic and chernozemic	sugar beets, barley carrots and wheat	not given
Bondietti et al. (1979)	see U	see U	leaves and stems	15
			fruit, vegetables or seed	14
	pot culture in greenhouse	floodplain soil	snapbeans	2
Dave et al. (1984)	see Pb	see Pb 2-15pCi ^{232}Th/g 5-62pCi ^{230}Th/g 2-38pCi ^{228}Th/g	grasses, legumes and tree species	8
Drichko and Lisachenko (1984)	field sampling of high-background farm soils	three soil classes 23-36 Bq ^{238}Th/kg	grain	22-72 samples
			potatoes	15-36 samples
Evans and Eriksson (1983)	see U	see U 34±21 Bq ^{232}Th/kg	mature rape, barley and wheat grain	8 fields, 2 isotopes
			as above, straw	as above
			as above, mid-season	as above
Furr et al. (1977)	see Pb	see Pb 2-5 µg Th/g	cabbage	see Pb
Garten et al. (1981)	see U	see U 20 µg Th/g	fescue	9 plant samples

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.0007 (-7.2±0.8)		see U
	0.042(-5.5±0.26)		Data for ^{228}Th, we averaged four values gigven by the authors, each original value was listed as ±0.001
	0.0028 (0.0001-0.006)		see U
	0.0002 (0.0001-0.002)		
	0.00003		Authors commented that soil volume was too small
	0.018 (-4.0±1.6) implied range to as low as 0.0016		Implied range the result of no detectable Th in some soils that contained 62pCi ^{230}Th/g
	0.005-0.008		We assume the results were expressed on a dry weight basis
	0.003-0.004		
	0.00047 (-7.7±1.2)		
	0.00089 (-7.0±1.0)		
	0.0005 (-7.6±1.4)		
	0.08		
	0.00025		

Table 3.2 Continued

reference	type of study	soil information	plant information	number of sites or observations
Graschenko et al. (1982)	radionuclides in agricultural fertilizers	soddy podzolic	wheat	not given
		chernozem	wheat	not given
		podzolic	barley	not given
		leached chernozem	barley	not given
		leached chernozem	peas	not given
		chernozem	rice	not given
Gruzdev and Rubtsov (1972)	field sampling	taiga, podzolic soils with sphagnum ~40 µg Th/g	birch, spruce and Ravan wood pulp	34 samples, GM of 10 means
			as above, bark	as above
			as above, leaves and needles	34 samples, GM of 8 means
		as above, ~150 µg Th/g	as above, wood pulp	34 samples, GM of 5 means
			as above, bark	as above
			as above, leaves and needles	34 samples, GM of 7 means
Laul et al. (1977)	field sampling of gardens	14 µg Th/g	corn, peas, squash and grass	1 sample of each crop
Moffet and Tellier (1977)	see U	see U, 5±3 µg Th/g	see U	see U
Mordberg et al. (1976)	see U	see U, 7.8 µg ^{232}Th/g	wheat grain	not given
			corn grain	not given
			buckwheat grain	not given
			dried peas	not given
			millet grain	not given
			sunflower seeds	not given

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
	0.024±0.004		Assumed to be on a dry weight basis, the authors did not specify whether these were GM or what measure of deviation was given
	0.006±0.001		
	0.028±0.005		
	0.009±0.002		
	0.032±0.004		
	0.005±0.001		
		0.37 (-1.0±0.9)	The authors believed that CR values decreased as soil Th concentrations increased
		0.51 (-0.67±0.88)	
		0.31 (-1.2±0.64)	
		0.071 (-2.6±1.0)	
		0.12 (-2.1±1.2)	
		0.041 (-3.2±0.77)	
	0.00057-0.0049		
	0.0011(-6.8±1.3)	see U	
0.0007±0.0002		see U	
0.0004±0.0014			
0.0026±0.001			
0.0002±0.001			
0.001±0.0012			
0.0003±0.0014			

Table 3.2 Continued

reference	type of study	soil information	plant information	number of sites or observations
Rayno et al. (1980)	see U	see U, up to 87pCi ^{230}Th/g	4 forage species	1
Titaeva et al. (1979)	see U	soddy meadow, 28 µCi ^{230}Th/g	grasses and forbs	not given
		1700 µCi ^{230}Th/g	grasses and forbs	
		gleyed podzol, 6400 µCi ^{230}Th/g	tree species	
Verkhovskaya et al. (1972)	field sampling	gleyed and podzolic soils	several native plant species	not given
Whicker and Ibrahim (1983)	see U	1-13 pCi^{230}Th/g	see U	see U

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
	0.11		see U
		0.041(-3.2±1.4)	see U
		0.0034(-5.7±2.1)	
		0.00011(-9.1±0.9)	
		about 1	
	0.01-0.14		

Table 3.3 Compilation of CR values for Pb

reference	type of study	soil information	plant information	number of sites or observations
Cloutier et al. (1985)	field sampling of tailings area	revegetated U tailings	grasses and trefoil	3
Dave et al. (1984)	field sampling of tailings area	revegetated U tailings 5-440 pCi/g	grasses	6
			legumes	6
			white birch stems	4
			white birch leaves	4
			trembling aspen stems	4
			trembling aspen leaves	4
Davey and Conway (1974)	see U	see U 0.6-5.9 µg Pb/g 1-5 µg Pb/g 5% loss on ignition	pasture grasses	3
			paperbark leaves	5
Dedolph et al. (1979)	field plots	field adjacent to a highway 27-82 µg Pb/g	radish leaves	4 replicates
			radish roots	as above
Dreesen and Marple (1979)	see U	see U 30-70 µg Pb/g	grass	6 soils (4-6 internal replicates)
			dryland shrub	6 soils (see above)
Furr et al. (1977)	pot culture 5 kg soil/pot	7% mixture of fly ash with soil 0.2-17 µg Pb/g	cabbage	15 fly ashes, 2 replicates
Kalin (1982)	field sampling of tailings area	U tailings	two native grass species	12

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.005		Plant content of stable Pb marginally above detection limits
	0.02 (-3.7±2.2)		Measurement of ^{210}Pb, actual concentrations in leaves were more uniform than were CR values, CR values were inversely proportional to soil concentrations (CR's of 0.1-0.7 when soil concentrations were less than 10pCi/g, and 0.001-0.02 when soil concentrations were above 200pCi/g).
	0.027 (-3.6±1.8)		
	0.023 (-3.7±1.6)		
	0.014 (-4.2±1.6)		
	0.005 (-5.3±2.1)		
	0.01 (-4.6±1.8)		
	0.25 (-1.4±0.8)		see U
	0.34 (-1.1±0.8)		see U
	0.18-0.24		Generally the CR values for leaves decreased with distance from the highway and CR values for roots increased, atmospheric deposition of Pb was appreciable near the highway.
	0.0085-0.030		
	0.3 (-1.2±0.7)		see U
	0.02 (-4.1±1.4)		see U
	1.07		stable Pb
	0.06		Stable Pb and ^{210}Pb gave the same CR values

Table 3.3 Continued

reference	type of study	soil information	plant information	number of sites or observations
Kalin and Sharma (1982)	field sampling of tailings site	waterlogged tailings 10-140 pCi ^{210}Pb/g	cattail roots (scrubbed and washed)	14 sampling points
Kovalevskii (1979)	field sampling, pre-mineralized area	not given	Triodia pungens, Eriache mucronata, Policarpa glabra, Tephrosia sp.	not given
Mahon and Mathewes (1983a)	see U	see U, 3.0±2.7 pCi/g	see U	see U
Mahon and Mathewes (1983b)	see U	see U, ~6 pCi/g	see U	see U
Milosevic et al. (1980)	see U	see U 23-270 mBq ^{210}Pb/kg	see U	8 sites
Moffet and Tellier (1977)	see U	see U, 36±60 pCi ^{210}Pb/g	red fescue grass	see U, 1 ratio
			3 other grass species	see U, GM of 3 ratios
		as above, 130±102 pCi ^{210}Pb/g	red fescue grass	see U, 1 ratio
			3 other grass species	see U, GM of 3 ratios
Rayno et al. (1980)	see U	see U, 1-200 pCi ^{210}Pb/g	see U	see U
Sharma and Kalin (1982)	field sampling	Elliot Lake tailings areas, 0.2-77 pCi ^{210}Pb/g	cattails	8 sites
			sedges	4 sites
			trembling aspen	3 sites
			white birches	7 sites
		Bancroft tailings areas, 18-190 pCi ^{210}Pb/g	cattails	4 sites
			sedges	4 sites
			trembling aspens	4 sites
			white birches	3 sites
		Uranium City tailings area, 24-370 pCi ^{210}Pb/g	foxtails	5 sites
			salt marsh grass	5 sites

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
	0.72(-0.33±.1.7)		
		0.062(-2.78±.95)	
	0.41(-0.90±0.77)		Measurements of ^{210}P, see U, authors believed deposition of Pb on foliage may have biased the values.
	0.069(-2.7±0.31)		Measurements of ^{210}P, lake water contained <0.5 pCi/g.
	0.069(-2.6±0.90)		see U
	0.19		Red fescue had significantly higher Pb concentrations, as did one of the five sites.
	0.017(-4.1±0.44)		
	0.080		
	0.0017(-6.4±0.20)		
	0.08-0.54		see U
	0.024(-3.71±.60) 0.35(-1.06±2.24) 0.18(-1.73±.96) 2.5(0.90±1.02)		Authors compared ^{210}Pb and stable Pb, we computed CR values only for ^{210}Pb
	0.084(-2.48±1.03) 0.082(-2.50±.79) 0.54(-0.62±.79) 0.39(-0.94±1.43)		
	0.033(-3.42±1.36) 0.038(-3.28±1.75)		

Table 3.3 Continued

reference	type of study	soil information	plant information	number of sites or observations
Sheard et al. (1986)	see U	see U, 0.03-1.05 Bq ^{210}Pb/g	12 tree and shrub species	96
Simon and Fraley (1986)	field, injected contamination	upwind of U mine fine loam pH 7.8 1.9% organic 57 kBq ^{210}Pb/g	native gramineae native phlox native chrysothamnus	duplicate plots sampled with time over two years
Tracy et al. (1983)	see U	8-550 pCi ^{210}Pb/g	leavy vegetables roots and stems fruit	6 samples 9 samples 6 samples
Whitehead et al. (1971)	field sampling	19-5000 pCi ^{210}Pb/g	4 native species	not given

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.99(-0.02±0.08)		The authors believed that deposition of Pb from the atmosphere may have contributed to the plant concentrations they observed.
	0.008		The authors computed GM's of CR values, the values decreased about five-fold from the first sampling to a steady-state level
	0.022		
	0.01		
0.001(-7.06±.71)			
0.0007(-7.77±1.12)			
0.0004(-8.14±.95)			
	0.055(-3.0±1.6)		

3.2 Selection of the Most-Relevant Data

Many of the data summarized in Tables 3.1 to 3.3 represent cases that are not especially appropriate for CR data for use near Canadian nuclear facilities. To be appropriate, the data should meet the following criteria:

1) be measured in the field, or at least outdoors in a realistic environment
2) be clearly independent of other contamination pathways such as atmospheric deposition, contaminated groundwater, contaminated subsoils (such as soil-covered tailings) etc.
3) be representative of soils, plants and environments typical of Canada.

We have selected the most-relevant data from Tables 3.1 to 3.3 and show these values in Tables 3.4 to 3.6.

3.3 Deficiencies of data

The categories we have chosen in Tables 3.4 to 3.6 reflect the amount of data we believe would be sufficient to assess dose consequences from U, Th and Pb. It is very clear that fev of these data are currently available. The most notably paucity of data is for food crops, especially for specified soil types. Categorization by soil type is an important refinement of CR data, especially when data are to be chosen to represent specific sites where soil-type information is available. The availability of relevant data decreases in the order U > Th > Pb.

Table 3.4 Categorization of CR values for U by soil and plant type for selected data[1]

Plant Type	Soil Type: Clay to Loam	Loam to Sand	Organic	Tailing	Not Specified
		native species			
trees	0.0007-0.01(1)[2]	0.002-0.05(4)	0.002-0.004(1)	-----	0.001-0.001(2)
shrubs	-----	0.002-0.03(4)	0.002(1)	-----	-----
annuals and grasses	0.01(1)	0.008-0.03(2)	-----	0.001-0.03(3)	0.001-0.008(2)
not specified	-----	-----	0.00001-0.0001(1)	-----	-----
		cultured species			
cereals	0.001-0.0083(3)	0.001(1)	-----	-----	0.001-0.003(2)
fruits	0.0002(1)	-----	-----	-----	0.003-0.01(3)
vegetables	0.0002(1)	-----	-----	-----	0.003-0.008(3)
root crops	-----	-----	-----	-----	0.002-0.007(3)
forage	-----	0.003(1)	-----	-----	-----
not specified	-----	-----	-----	-----	-----

[1] These data were selected as field-derived data that had no stated or implied confoundment by other processes such as atmospheric deposition, contaminated groundwaters, etc.

[2] The CR values represent means or ranges as shown in Table 3.1. Conversion to a dry weight basis was made when necessary by assuming the dry matter contents to be 90% for grains, 20% for vegetables and root crops and 5% for fruits. Numbers in parentheses are the number of reference sources. The reference sources used are listed in Table 3.7.

Table 3.5 Categorization of CR values for Th by soil and plant type for selected data[1].

Plant Type	Soil Type: Clay to Loam	Loam to Sand	Organic	Tailing	Not Specified
			native species		
trees	-----	-----	0.002-0.02(1)	-----	10^{-6}(1)
shrubs	-----	-----	-----	-----	-----
annuals and grasses	0.003(1)	-----	-----	0.001-0.02(2)	0.0001-0.001(1)
not specified	-----	-----	-----	-----	0.03(1)
			cultured species		
cereals	0.0007-0.042(3)	0.0005-0.02(3)	-----	-----	0.0002-0.003(1)
fruits	0.00003(1)	-----	-----	-----	-----
vegetables	0.0002-0.04(2)	-----	-----	-----	0.0006(1)
root crops	-----	0.003(1)	-----	-----	-----
forage	-----	-----	-----	-----	-----
not specified	-----	-----	-----	-----	-----

[1] These data were selected as field-derived data that had no stated or implied confoundment by other processes such as atmospheric deposition, contaminated groundwaters, etc.

[2] The CR values represent means or ranges as shown in Table 3.2. Conversion to a dry weight basis was made when necessary by assuming the dry matter contents to be 90% for grains, 20% for vegetables and root crops and 5% for fruits. Numbers in parentheses are the number of reference sources. The reference sources used are listed in Table 3.7.

Table 3.6 Categorization of CR values for Pb by soil and plant type for selected data[1].

Plant Type	Soil Type: Clay to Loam	Loam to Sand	Organic	Tailing	Not Specified
		native species			
trees	-----	-----	-----	0.005-0.03(1)	-----
shrubs	-----	-----	-----	-----	-----
annuals and grasses	0.008-0.01(1)	-----	-----	0.002-0.08(4)	-----
not specified	-----	-----	-----	-----	.06(1)
		cultured species			
cereals	-----	-----	-----	-----	-----
fruits	-----	-----	-----	-----	0.0001(1)
vegetables	-----	-----	-----	-----	0.0003(1)
root crops	-----	-----	-----	-----	0.0002(1)
forage	-----	0.069(1)	-----	-----	-----
not specified	-----	-----	-----	-----	-----

[1] These data were selected as field-derived data that had no stated or implied confoundment by other processes such as atmospheric deposition, contaminated groundwaters, etc.

[2] The CR values represent means or ranges as shown in Table 3.3. Conversion to a dry weight basis was made when necessary by assuming the dry matter contents to be 90% for grains, 20% for vegetables and root crops and 5% for fruits. Numbers in parentheses are the number of reference sources. The reference sources used are listed in Table 3.7.

Table 3.7 Reference sources used in Tables 3.4, 3.5 and 3.6

category		references
	for Table 3.4, Uranium	
trees:	clay to loam	Garten (1980)
:	loam to sand	Mahon and Matheves (1983a), Sheard et al. (1986), Sheppard and Thibault (1983), Walker (1979)
:	organic	Sheppard and Thibault (1983)
:	not specified	Lopatkina et al. (1970), Titaeva et al. (1979)
shrubs:	loam to sand	Cannon (1952), Mahon and Matheves (1983a), Sheppard and Thibault (1983), Walker (1979)
:	organic	Sheppard and Thibault (1983)
annuals:	clay to loam	Garten et al. (1981)
:	loam to sand	Cannon (1952), Mahon and Matheves (1983a)
:	tailing	Amiro and Sheppard (1987), Moffet and Tellier (1977), Rumble and Bjugstad (1986)
:	not specified	Rumble and Bjugstad (1986), Titaeva et al. (1979)
not specified:	organic	CBCL Limited (1985)
cereals:	clay to loam	Arkhipov et al. (1985), Evans and Eriksson (1983), Mordberg et al. (1976)
:	loam to sand	Evans and Eriksson (1983)
:	not specified	Frindik (1985), Mordberg et al. (1976)
fruits:	clay to loam	Bondietti et al. (1979)
fruits:	not specified	Frindik (1985), Mordberg et al. (1976), Tracy et al. (1983)
vegetables:	clay to loam	Bondietti et al. (1979)
vegetables:	not specified	Frindik (1985), Mordberg et al. (1976), Tracy et al. (1983)
root crops:	not specified	Frindik (1985), Mordberg et al. (1976), Tracy et al. (1983)
forage:	loam to sand	Milosevic et al. (1980)
	for Table 3.5, Thorium	
trees:	organic	Gruzdev and Rubtsov (1972)
:	not specific	Titaeva et al. (1979)

annuals:	clay to loam	Garten et al. (1981)
:	tailings	Dave et al. (1984), Moffet and Tellier (1977)
:	not specified	Titaeva et al. (1979)
not specified:	not specified	Verkhovskaya et al. (1972)
cereals:	clay to loam	Arkhipov et al. (1985), Ashkinazi (1982), Evans and Eriksson (1983)
:	loam to sand	Drichko and Lisachenko (1983), Evans and Eriksson (1983), Graschenko et al. (1982)
:	not specified	Mordberg et al. (1976)
fruits:	clay to loam	Bondietti et al. (1979)
vegetables:	clay to loam	Ashkinazi (1982), Bondietti et al. (1979)
:	not specified	Laul et al. (1977)
root crops:	loam to sand	Drichko and Lisachenko (1983)

for Table 3.6, Lead

trees:	tailings	Dave et al. (1984)
annuals:	clay to loam	Simon and Fraley (1986)
:	tailings	Cloutier et al. (1985), Dave et al. (1984), Kalin (1982), Moffet and Tellier (1977)
not specified:	not specified	Whitehead et al. (1971)
fruits:	not specified	Tracy et al. (1983)
vegetables:	not specified	Tracy et al. (1983)
root crops:	not specified	Tracy et al. (1983)
forage:	loam to sand	Milosevic et al. (1980)

3.4 Comparison to Established Default Values

Several groups have attempted to compile or generate CR data for use in environmental impact studies, and many other agencies have adopted these values. Typically, the values are proposed as default values, recognizing that site-specific data would be the most meaningful. Two of the major compendiums, that of the Associate Nuclear Services in the United Kingdom (Coughtry et al., 1985) and that of the International Union of Radioecologists of the EEC (IUR, 1984) did not include U, Th or Pb in their compilations. The compilation for the United States Environmental Protection Agency (Baes et al. 1984) did include these elements. They selected CR values for vegetative tissue of 0.0085 for U (based on three references), 0.00085 for Th (based on two references and set an order of magnitude lower than U) and 0.045 for Pb (based on nine unspecified references). They selected corresponding CR values for reproductive tissues of 0.004, 0.000085 and 0.009, respectively, based on the same references. The Canadian Standards Association (CSA 1984) adopted the first set of values proposed by Baes et al.. Cassaday et al. (1985) chose CR values for U, Th and Pb for the Canadian National Uranium Tailings Program based partially on these compendiums and partially on original references.

Our compilation in Tables 3.1 to 3.3 includes, as far as we can tell, all of the data used by Baes et al. and a good many more references. For that reason, the values proposed by Baes et al. fall within the ranges we found in the literature.

4. SUMMARY AND CONCLUSIONS

This literature review was undertaken in response to a perceived need for more data and a better understanding of the uptake of U, Th and Pb by plants from soil. Among radionuclide elements of importance to the nuclear industry as a whole, isotopes of these elements have received less research attention. This is probably due to their relatively lov mobility in the environment and an emphasis on elements released into the environment by reactors, reprocessing plants, etc. However, in the vicinity of U mines and mills, U_{nat}, ^{230}Th and ^{210}Pb are

potentially important contributors to radioactivity in the environment.

This report is divided into three main sections. The first is an overview of the importance and characteristics of the concentration ratio (CR) model used to estimate uptake of radionuclides by plants. The underlying mechanisms and processes encompassed by the model are briefly described. The second section reviews what is known about the behaviour of U, Th and Pb in soils and plants, with some of the key points illustrated by data from the literature. The third section is a compilation of much of the CR data available in the literature for U, Th and Pb. Many data are available. However, on critical review of the appropriateness of the data in terms of experimental methods and types of soils and crops, a large proportion of the data must be rejected or only provisionally accepted. We have selected and presented data that fit our acceptance criteria. In general, there are substantial gaps in the information, especially for edible plants. The data are not sufficient to categorize CR values by soil type. The paucity of data is most extreme for Th and Pb. We recommend that experimental data be obtained to increase the amount of data available for edible plants on a range of soil types. We also recommend that these data be supported by information of the soil chemistry and plant growth rates, as these underlying factors have not been adequately reported by most studies in the literature.

5. RECOMMENDATIONS

5.1 General Recommendations

Maintain an open file of CR data so that new data can be incorporated as it becomes available.

Experimentally determine CR values for U, Th and Pb for edible crops on a range of soil types.

Support all new CR data with information on soil types, solubility of the elements in soil, plant growth rates, etc.

Statistically evaluate CR data to determine which values can be averaged and which should be compiled separately.

5.2 Recommendations for Experimental Methodology

Conduct experiments outdoors in soil containers of sufficient size to represent field conditions.

Uniformly mix radionuclides in the top 0-20 cm of soil.

Culture plants in terms of planting date, fertilizer level, water supply and harvesting dates to conform with common practice.

Report a complete characterization of the soils including pH, texture, organic matter content, carbonate content, and pore water chemistry.

Report measures of radionuclide solubility in the soils.

Report a complete characterization of the plants including estimates of growth rate at various stages, final yield, morphology and biomass and ash content partitioning among plant parts.

5.3 General Recommendations Not Specific to CR Data

Measure the radionuclide sorption characteristics on a large number and broad range of Canadian soils since these data (as K_d values) are themselves important to the modelling of radionuclide migration in the environment and are very helpful in predicting soil-specific CR values.

Evaluate a multivariate approach to the analysis of environmental transfer factors so that the interrelationship of the behaviours of radionuclides in soils, plants and animals is recognized and is made of use in selecting data for a specific case.

Evaluate the impact of non-uniform and subsurface contamination of soil on the uptake of radionuclides by plants.

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APPENDIX A

Technical audit trail

The intent of this appendix is to document the assumptions, procedures and detailed sources of data used to compute the CR values given in Tables 3.1 to 3.3. It will serve as a partial form of quality assurance. There is an entry for each reference we examined, including those identified in the computerized literature search but found to be unsuitable for inclusion in the Tables. The table, figure and page numbers given in each entry are those of the original manuscripts we obtained. Our general procedures were as follows:

- all averaging of CR values was done on $\log_e$-transformed data, regardless of whether raw data points or means computed by the original authors were averaged

- soil or plant concentrations were computed as arithmetic means

- the GM and GSD were presented if possible, ranges were presented where very few data were available, and single values were presented when estimates of variation were not available

- various CR values from a given reference were averaged when 1) the original authors had demonstrated no statistically significant differences among them, 2) in the absence of relevant statistical tests, the values were relatively similar ($\pm$ 1 GSD) or 3) the classification used by the original authors had no practical value in this review (for example, classification by cultivar for foreign cultivars is of no use for Canadian applications)

- soil concentrations from the top 0-20 cm were used to compute CR, although note was made if the authors had reported elevated concentrations and root penetration lower in the soil profile

- conversion between fresh weight, dry weight and ash weight modes of expression was done using data from the corresponding papers for ash and dry matter contents.

Adams et al. (1975)	- methods and soil data on page 8 - CR data from Table 8
Amiro and Sheppard (1987)	- CR data from caption of Fig. 1.
Arkhipov et al. (1985)	- plant concentrations taken as the average of 1977 and 1978 from Table 2 - soil concentrations from Table 1 - GSD reflects variance among the five treatment means
Armand and Landegren (1960)	- no acceptable CR data, substrate for peat mosses was water
Ashkinazi (translation dated 1982)	- CR values taken from Table 1
Baeva et al. (1975)	- CR data taken from the English abstract
Boileau et al. (1985)	- no acceptable CR data because the authors used a solution culture method
Bondietti et al. (1979)	- CR data for field study taken from Fig. 5.2 - CR data for pot culture taken from Table 5.9

Cannon (1952)	- CR data where n=11 based on rock analysis, not soil, calculated from Table 1 - other CR data calculated from Tables 6, 7 and 8 selecting data where soil-like materials were described as the substrate or where roots entered fissures in rock - the specific soil/plant pairs chosen were S7/P20, P22, P24; S8/P26; S20/P38; S22/P217; S28/P223; S11/P49, P52, P54; S13/P67, P69, P71; S12/P58; S14/P73; S1/P9,P10, P11, P12; Wamsutter sites.
CBCL Limited (1985)	- CR values taken from Table 22
Cloutier et al. (1985)	- CR for Pb from Elliot Lake tailings computed from data in Fig. 6
Dave et al. (1984)	- CR values computed from Table 1
Davy and Conway (1974)	- CR values for pasture grasses computed from Table 24 - CR value for palm fruit computed for the Magela Channel sample in Table 25 and using the mean soil concentration from Table 16 - CR values for paperbark leaves computed from Table 26 using the mean soil concentration on a dry weight basis from Table 19 - remainder of data from Davy and Conway was not used because of difficulties in pairing plants with soils and because of an unclear role of flooding as a source of nuclide in the plants
Dedolph et al. (1979)	- CR values computed from soil and plant concentrations given in Table II

Dreesen et al. (1978)
- no acceptable CR data because U concentrations in plants considered by the authors to be due to surface contamination by blown soil and tailings materials.

Dreesen et al. (1982)
- CR values for the greenhouse study taken from Table V

Dreesen and Cokal (1984)
- no soil concentrations were given to be able to complete CR values

Dreesen and Marple (1979)
- soil concentrations were derived for each treatment by averaging the concentrations of the soils and tailings weighted by the corresponding weight in each 2.6-L pot; the soil weights were taken from Fig. 1 and the concentrations from Table 1
- CR values were computed from the plant concentrations given in Tables 2 and 3

Drichko and Lisachenko (1983)
- CR values were taken from Table 3

D'Souza and Mistry (1970)
- CR values were not compiled because authors used a solution culture technique

Dunn (1981)
- soil concentrations from Table 2 were averaged, weighting the values by the number of samples given in Table 1, an overall weighted mean of 4.7 µg U/g was used
- plant concentrations from Table 2 were averaged to give the groups we present, weighting as above

- the CR values on a overall basis were computed from these weighted means
- the CR values on a dry weight bases were computed using the ash contents for each plant part given on pages 440-441

Dunn (1983)
- CR values on an ash basis computed from Table 22.5 as twig ash / B_r soils for each pair
- mean CR on a dry weight basis used the ash content of black spruce twigs reported by Dunn (1981)

Evans and Eriksson (1983)
- CR values computed using the values given in Tables 21 and 22 for HCl extractions of soil and corrected for the extraction efficiency as given in Table 24 (this is the procedure used by the authors to compute CR from their data)

Fedorov and Romanov (1969)
- may have relevant data for U but the translation (and apparently the original Russian manuscript) is not comprehensible

Frindik (1985)
- CR values taken from Table 4 averaging the values for each crop type, the ranges given in Table 4 for cereals were presented and the GSD for vegetables and fruits reflect the species to species variability in Table 4

Furr et al. (1977)
- the GM of ash Pb and Th concentrations for the 15 ashes used in the plant experiment computed from Table II
- the concentration in soil computed as 7% of that in ash because a 7% ash/soil mixture was used to grov the plants

- the GM of the plant Pb and Th concentrations computed from the ranges given on page 1200
- the CR computed from the above GM concentrations

Garten (1978)
- ranges of CR values on page 445 reported in lieu of actual references by Hanson and Miera (1976) and Gilbert et al. (1976)
- the values are dry weight basis as defined in the footnote on page 443

Garten (1981)
- CR values derived from Table 3

Garten et al. (1981)
- CR values computed from plant and soil concentrations that were computed from the element ratios given in Fig. 2

Garten (1980)
- CR ranges from Table 1 including both ^{234}U and ^{238}U
- soil information from Garten et al. (1981)

Grashchenko et al. (1982)
- CR values taken from Table 1, assumed to be on a dry weight basis

Gruzdev and Rubtsov (1972)
- CR values on ash weight basis averaged from means given in Table 2
- CR data for substrate concentrations listed as <50 and 50-100 µCi/h (these units were as the translator interpretted them) were averaged together to give values for the ~40µg Th/g class

Gulati et al. (1980)
- CR values for mature plants computed from Table 1 by first subtracting the concentration of U in control plants from those receiving 1.5, 3 or 6 µg U/g

	- the soil U concentration taken as the concentrations applied
Kalin (1982)	- CR values computed from Table 19
Kalin and Sharma (1982)	- CR values taken from Figure 5a
Kovalevskii (1979)	- CR values taken from Table 7 (chapter 2)
Laul et al. (1977)	- CR values computed for four samples from concentrations given in Table 1
Lopatkina et al. (1970)	- CR values from Tables 1 and 2 averaged into 3 classes based on concentration in soil
Mahon and Matheves (1983a)	- CR values from Table 8 averaged, weighting the log-transformed mean in Table 8 by the corresponding sample size, resulting in a GM representing 51 samples
Mahon and Matheves (1983b)	- CR values from Table 5, weighted as above resulting in a GM representing 12 samples
Milosevic et al. (1980)	- soil concentrations for each site computed as the average of the 0-5, 5-10 and 10-15 cm depths from Table 4 - CR values computed for each site using the plant concentrations given in Table 3
Moffet and Tellier (1977)	- concentrations in soil from Table 2 and in plant from Table 4

- the decision of sites and plant species to average together based on t-tests shown in Table 2 and Table 3

Mordberg et al. (1976) - CR values from Table 4, they are equivalent to the slopes of the equations n=bx in Table 2

Morishima et al. (1976) - CR values taken from Table 3, the ash weights given must be incorrectly labelled and are incompletely described

Morishima et al. (1977) - fresh weight CR values taken from Table 5 for the Nara region only, because the concentration of U in the irrigation water was lowest for this region
- ash weight CR values computed by averaging the U concentration in plant ash in Table 1 for each plant type, and dividing by the soil concentration from Table 4

Prister and Prister (1970) - CR values computed from Table 1, they appear to have been measured in an overly artificial system

Rayno et al. (1980) - CR values taken from Table 3

Rumble and Bjugstad (1986) - CR values taken from Table 4

Saumande et al. (1982) - the authors measure alpha and beta activities only in field samples and thus did not differentiate radionuclides

- the artificial experiment with U was hydroponic and therefore the CR values were not acceptable

Schreckhise and Cline (1978) - data taken from the abstract as only the abstract was available

Sharma and Kalin (1982) - CR values were computed for each sample as the ratio of concentrations in "leaves" to that in "root region tailings", from Tables 1, 2, 3, 4, 6 and 7

Sheard et al. (1986) - CR data for U calculated as individual pairs from plant and soil data in Table 21-23
- CR data for Pb as above using Tables 11-13

Sheppard et al. (1982) - plant concentrations were less than 3 µg U/g and soil concentrations were as high as 40 µg U/g, thus CR values were less than 0.07

Sheppard and Thibault (1983) - CR values on ash/ash weight basis taken from Table 2
- mean CR from Table 2 converted to dry weight basis using mean soil ash content from Sheppard and Thibault (1983) and mean plant ash contents for each plant type from Sheppard and Thibault (1984)

Sheppard et al. (1984) - soil concentrations for the sand soil for the top 15 cm at the end of the experiment were estimated from Fig. 2 and 3
- plant concentrations were computed from total U uptake (Table 4) divided by mean plant weight (Table 2)

Sheppard et al. (1985) - CR values taken from Table 3

Sheppard and Evenden (1985) - soil concentration computed as averages for the top 12 cm of soil from original data provided by the authors
- CR values computed for individual lysimeters and averaged across those treatments which did not affect the CR values

Smith et al. (1982) - CR values computed from Tables 10 and 11 for each isotope, and then averaged
- an errata for Table 10 indicated the correct units were fCi/g

Titaeva et al. (1979) - soil concentrations taken from Table 1 and used to compute CR values for each species in Table 3

Tracy et al. (1983) - CR values computed for individual plant samples in Table 2 (Pb) and 3 (U) using the soil concentration for the 0- to 15-cm depth

Van Netten and Morley (1981) - CR values averaged from Table 1

Van Netten and Morley (1982) - CR values averaged for stalks and seeds from Table 1, excluding data for soil A as there is an error in either the CR values or the soil concentrations given by the authors

Van Netten and Morley (1983) - CR values averaged from Table 1

Verkhovskaya et al. (1972) - CR values taken from text on page 4 of translation

Walker (1979)	- weighted average of soil concentration computed from the medians given in Table 3 and assuming the depth of the B horizon was twice that of the combined A_o and A_1 horizons - CR values computed from medians in Table 3 for needle, twig, cone, leaf, stem and seed tissues.
Whicker and Ibrahim (1983)	- CR values taken from Table IV.8 for the natural background and reclaimed areas only
Whitehead et al. (1971)	- CR values were computed from concentrations given in Table 1 for higher plants
Yamamoto et al. (1968)	- soil concentrations taken from Table 3 and plant concentrations from Table 4 - we computed the GM of six means
Yliruokanen I (1975)	- plant concentrations of U, Th and Pb are given but the only substrate analysed was a mineralized rock outcrop near where the plants were collected - we did not complete CR values because the root zone was not analyzed

APPENDIX B

Compilation of Data from Russian translations

Several of the reports listed in the bibliography could only be obtained in original Russian. These reports were translated and the data appended here when the translations were completed. A separate technical audit trail follows the Appendix tables.

Table B.1 Appendix to compilation of CR values for U

reference	type of study	soil information	plant information	number of sites or observations
Arkhipov et al. (1984)	greenhouse large pots	podzolic 35 Bq/kg	barley grain	3 reps x 3 soils
		chernozem	barley plant	3 reps x 3 soils
		brown and sierogem	barley grain	3 reps x 2 soils
Arkhipov et al. (1985)	field plots	dark chestnut 11 Bq/kg	grass	4 reps x 3 treatment
			barley grain	
Baeva et al. (1975)	field	subtropical region	10 species	
Bufatin et al. (1986)	greenhouse	flooded chernozem 100 mg ^{238}U/kg 500 mg ^{239}U/kg	rice grain	
		1000 mg ^{238}U/kg		
Mordberg et al. (1977)	chernozem 2-40 mg/kg	potatoes and beets cabbage tomato and cucumber pumpkin and melon		
Nikolaev & Konovalova (1980)	field	not given	not given	

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.02		
	0.017		
	0.013		
	0.055		
	0.014		
	0.035 - 0.186		
	0.0018		
	0.0014		
	0.0009		
0.1 - 3.0	0.4 - 12	16 - 59	
0.6	12		
0.3 - 0.7	6 - 14	63 - 180	
0.3 - 2	6 - 40	100 - 670	
	0.001 - 0.1		

Table B.2 Appendix to compilation of CR values for Th

reference	type of study	soil information	plant information	number of sites or observations
Arkhipov et al. (1984)	see U	podzolic 60 Bq/kg	barley grain	see U
		chernozem	barley grain	
		brown or sierozem	barley grain	
Arkhipov et al. (1985)	see U	see U 3 Bq/kg	grass	see U
			barley grain	
Mordberg et al. (1977)	see U	see U 4-46 m Ci/kg	potatoes and beets	
			tomatoes and cucumbers	
			pumpkin and melon	
Nikolaev & Konovalova (1980)	see U	not given	not given	
Volkova (1978)	field	podzolic	potatoe and beets	
Volkova et al. (1980)	field	podzolic	barley grain	not given

Concentration Ratio (CR)			Comments
fresh weight basis	dry weight basis	ash weight basis	
	0.001 - 0.005		
	0.001		
	0.001		
	0.14		
	0.004		
0.1 - 2	0.4 - 8	8 - 300	
0.4	8	90	
0.04 - 0.5	0.8 - 10	13 - 170	
	0.007 - 0.4		
	0.012 (-4.4 ± 0.5)		
	0.012 (-4.4 ± 1.1)		

Table B.3 Appendix to compilation of CR values for Pb

reference	type of study	soil information	plant information	number of sites or observations
Arkhipov et al. (1984)	see U	8 soils 3000 Bq/kg	barley grain	3 reps x 8 soils
Arkhipov et al. (1985)	see U 59-1110 Bq/kg	grass barley grain	see U	
Mordberg et al. (1977)	see U	see U 2-11 Bq/kg	potatoes and beets 6 species	

Concentration Ratio (CR)			
fresh weight basis	dry weight basis	ash weight basis	Comments
	0.001		
	0.23		
	0.011		
0.2 - 6.2	0.8 - 25	30 - 490	
3	50	700	

Technical Audit Trail for Appendix B

Arkhipov et al. (1984) - CR data taken from Table 2.

Arkhipov et al. (1985) - data for the control treatments of no irrigation and furrow irrigation averaged and data for furrow irrigation with ^{226}Ra - enriched waters used alone, data for sprinkler irrigation not used

- soil concentrations taken from Table 1 and plant concentrations from Table 2.

Baeva et al. (1975) - range of values of CR for ten species taken from text (page 66 or original Russian, page 4 of translation).

Bufatin et al. (1986) - CR values taken from text (page 98 of original Russian and pages 8 and 9 of translation).

Mordberg et al. (1977) - CR values in Table 2 converted to dry and ash weights using moisture and ask contents given in Table 2.

Nikolaev and Konovalova (1980) - range of CR taken from text (page 53 of original Russian, page 3 of translation).

Volkova (1978) - CR data taken from Table 1

Volkova et al. (1980) - data for the field experiment computed from Table 2, computing the GM of the three treatments.